Appendix A
A Toxicological Review of the Ozone NAAQS

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The Texas Commission on Environmental Quality (TCEQ) strives to protect our state's public health and natural resources consistent with sustainable economic development. To accomplish this mission, we base decisions on the law, common sense, sound science, and fiscal responsibility and we strive to ensure that regulations are necessary, effective, and current. In accordance with this mission, the TCEQ agrees with EPA that the NAAQS for ozone should protect public health. However, we would like to emphasize that modeling presented in the Health Risk and Exposure Assessment (HREA) for ozone indicates a lower standard may result in additional premature mortality for some areas of the country, including Houston (figures 7B-2, 7B-4, and appendix 7).

EPA presented the information below in black text in Chapter 7 of the HREA, i.e. the core analysis. We added the red text, summing the total impact of ozone reductions on mortality in Houston to reflect the information presented in Appendix 7, but not the core analysis. According to EPA's modeling, the net result will be an ***increase*** in mortality in Houston for any of the alternative standards under consideration.

Number of Premature Mortalities Predicted by EPA to Occur in Houston		
	Presented by EPA in Chapter 7	Based on Full Analysis found in Appendix 7
Meeting Current Standard (75 ppb) from Present Day (2007) Ozone Levels	Not presented	47 More
Going from 75 ppb to 70 ppb	1 more	48 More
Going from 75 ppb to 65 ppb	3 Less	44 More
Going from 75 ppb to 60 ppb	12 Less	35 More
2009 Data from Final draft of HREA		

In addition, when considering alternative O_3 standards (60-70 ppb), the lower end of the proposed range is not well-supported. In fact, EPA states in its Policy Assessment (PA) for

ozone that at lower concentrations "...the likelihood and magnitude of a response becomes increasingly uncertain..." (PA p3-1) and elsewhere that the "...the relative importance of background O_3 would increase ...with a lower level of the O_3 NAAQS" (PA p2-27).

Our conclusions are summarized below and are based on a thorough evaluation of the relevant scientific literature as well as the analysis presented by EPA in its HREA and PA for ozone. This evaluation was conducted by our Toxicology staff, which consists of 9 PhD and 5 Master's level scientists, including 3 certified by the American Board of Toxicology, and collectively representing over 150 years of experience. These individuals specialize in toxicology and risk assessment related to air contaminants. The methods employed by our scientists are state of the science, peer reviewed, and published. These guidelines and resulting toxicity factors are used by several other states as well as other countries, including Canada, Australia, Israel, Taiwan, Austria, Belgium, Mexico, China, and the Netherlands. While the EPA sets NAAQS standards for 6 pollutants, the TCEQ establishes acceptable levels for thousands (>5000) of air contaminants.

It is our determination, based on an evaluation of the available scientific information, that EPA has not made the case that a lower standard will improve public health, especially in light of their prediction of ***increased*** mortality as a result of ***lower*** ozone levels, as shown in the table above.

A thorough weight of evidence is lacking in EPA's analysis.

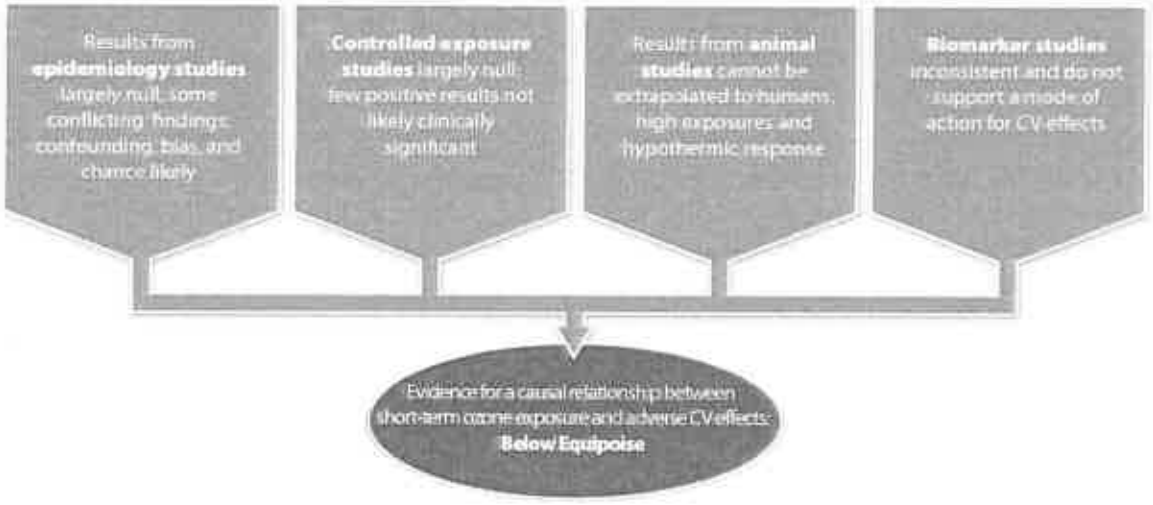
It is not clear how EPA has applied its weight of evidence framework to integrate results from human clinical studies, epidemiological studies, and animal studies. Throughout the draft document, studies are described as "positive" without indicating whether the results were statistically significant, biologically plausible, clinically meaningful, or consistent with other studies. For example, it is not clear how newer studies (Smith *et al.* 2009, Zanobetti and Schwartz 2008, and Jerrett *et al.* 2009) were weighed against other studies that reported "small associations or no associations" between ozone and mortality. In its consideration of weight of evidence, it is not clear how EPA evaluated consistency across studies or whether evidence evaluated across realms was ultimately considered.

A rigorous weight of evidence evaluation should be conducted, rather than giving positive results more weight than null results simply because they are positive. Based on EPA's incomplete evaluation of the evidence, it is not clear that there are causal relationships for health effects at ozone exposures below the current standard. EPA should use a rigorous weight of evidence as recommended by the National Academy of Sciences (NAS), and should not make policy judgments without assessing all of the available evidence.

The draft HREA uses endpoints previously determined to have "Suggestive," "Likely Causal" as well as "Causal" relationships with ozone exposure. Only endpoints with sufficient evidence to indicate a causal association should be used in setting a NAAQS. Therefore, only respiratory endpoints that can be demonstrated to be caused by short-term exposure to ozone should be used. It is especially problematic to use mortality supposedly related to long-term exposure to ozone as this was categorized as merely "Suggestive" in the ISA and lacks adequate evidence from scientific literature to be utilized in setting a standard. EPA should select endpoints that have clear biological plausibility and clinical significance.

The selection of endpoints is inappropriate in some cases, e.g. cardiovascular effects.

In the 2013 ISA, EPA stated that the epidemiology evidence for cardiovascular endpoints is inconsistent and lacks coherence across realms of evidence. Although EPA determined that the evidence was sufficient to conclude a "likely" causal relationship between short-term ozone exposure and cardiovascular-related endpoints, a thorough review of the literature reveals mixed results. In a soon-to-be-published report, Goodman *et al.* (2014) rigorously evaluated the studies reviewed by EPA as well as additional available literature. The authors reviewed 90 epidemiology studies, 8 controlled human exposure studies, 11 experimental animal studies, and 26 biomarker studies. Using a systematic weight of evidence approach it was determined that the available studies reported mixed results with positive, null and negative associations being reported. There was no consistent evidence that ozone affects biomarkers of inflammation, coagulation, oxidative stress, lipids and glucose metabolism, or overall cardiovascular health. There was no consistency in effects (e.g. increase versus decrease) and effects were often in the opposite direction of an adverse effect. Based on this information, the mode-of-action data do not support a biologically plausible mechanism for cardiovascular effects of ozone. Taken together, the weight of evidence (see figure below) indicates that a causal relationship between short-term exposure to ambient ozone levels and adverse effects on the cardiovascular system is not likely in humans. Put another way, while a few studies might have found a correlation between ozone concentrations and heart attacks, the overall weight of the evidence does not lead to the conclusion that low-level ozone exposure causes heart attacks. These results indicate that there is not adequate evidence of a causal relationship and therefore cardiovascular endpoints should not be included in the PA.



Lung function decrements are not likely to be adverse.

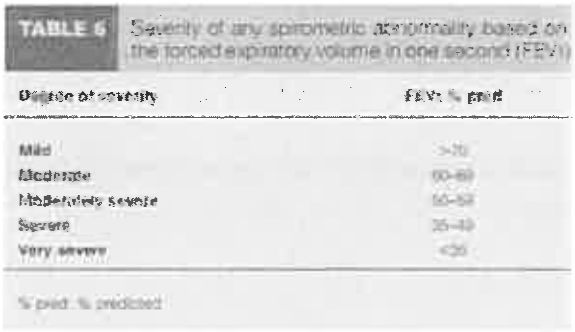
The EPA has selected hypothetical lung function decrements over specific cutoff values (≥10%, 15%, or 20%) in one year. However, determining the percent or number of individuals that experience at least one hypothetical FEV_1 decrement over a particular cutoff likely overestimates the significance of individual responses, particularly at lower ozone exposure levels because of the individual variability of FEV_1 when measured by spirometry. Indeed, Pellegrino *et al.* 2005 noted that FEV_1 decrements can vary by as much as 5% in healthy adults within a single day and by 15% or more from year to year. Moreover, this same study noted that changes in FEV_1 correlate "poorly with symptoms and may not, by itself, accurately predict clinical severity or prognosis for individual patients." In addition, because the selected model estimates individuals with at least one hypothetical lung function decrement over each of the cutoffs, it is possible that many of the selected individuals have only a single occurrence of effect which is of questionable clinical significance.

The draft HREA does not accurately reflect the available data addressing the selected lung function endpoint of FEV_1 decrements. The low concentration studies by Adams *et al.* (2002 and 2006), Schelegle *et al.* (2009), and Kim *et al.* (2011) all indicate a threshold below 70 ppb at which there are no statistically significant adverse effects associated with ozone. EPA should explain its rationale for modeling risks below 70 ppb ozone levels when controlled human exposure studies do not indicate effects at these exposure levels.

In the HREA, EPA describes the exercise patterns in the clinical studies examining lung function as "moderate" when individuals exercised 50 minutes of each hour for a prolonged period of 6.6 hours. However, as noted in Folinsbee *et al.* 1988 and McDonnell *et al.* 1991, this simulates work performed during a day of heavy manual labor in outdoor workers. In fact, exercise at this level for 6 to 8 hours should be considered as "heavy" or "strenuous" instead. We would like to point out that CASAC commented on this in the first draft HREA, saying the clinical studies cited by EPA used "...unrealistic elevated minute ventilations" and that "overall ventilations are ≥ mean ventilations that might be encountered during a day of heavy severe manual labor and represents the higher end of ventilations that might be encountered in the normal population for this prolonged period (6.6 h)."

Finally, EPA has focused much of its attention on small hypothetical changes in FEV_1. Other endpoints, such as respiratory symptoms, are generally required to determine if an individual is truly experiencing an adverse effect. In fact, the American Thoracic Society (2000) guidelines for identifying adverse effects link pulmonary changes with respiratory symptoms, clearly stating that reversible loss of lung function *in conjunction with* symptoms should be considered adverse. Thus, while FEV_1 may be a useful and sensitive biomarker, taken alone, it likely overestimates the number of individuals experiencing adverse effects. In addition, these lung function decrements would be transient, reversible, would not interfere with normal activity and would not result in permanent injury or respiratory dysfunction (Goodman *et al.* 2013).

The table below was taken from Pellegrino *et al.* 2005, an American Thoracic Society/European Respiratory Society joint publication, describes severity of FEV_1 decrements. Note that a change in FEV_1 of less than 30% is considered mild. This is in contrast with EPA's analysis which designates changes in FEV_1 of 10% as adverse.

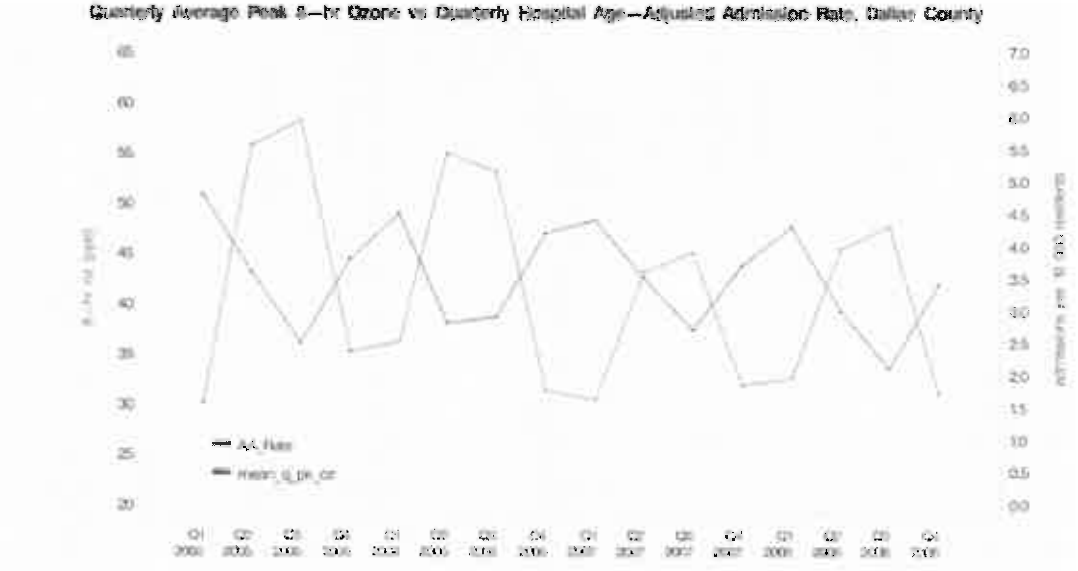


The evidence for ozone-caused new-onset asthma is insufficient.

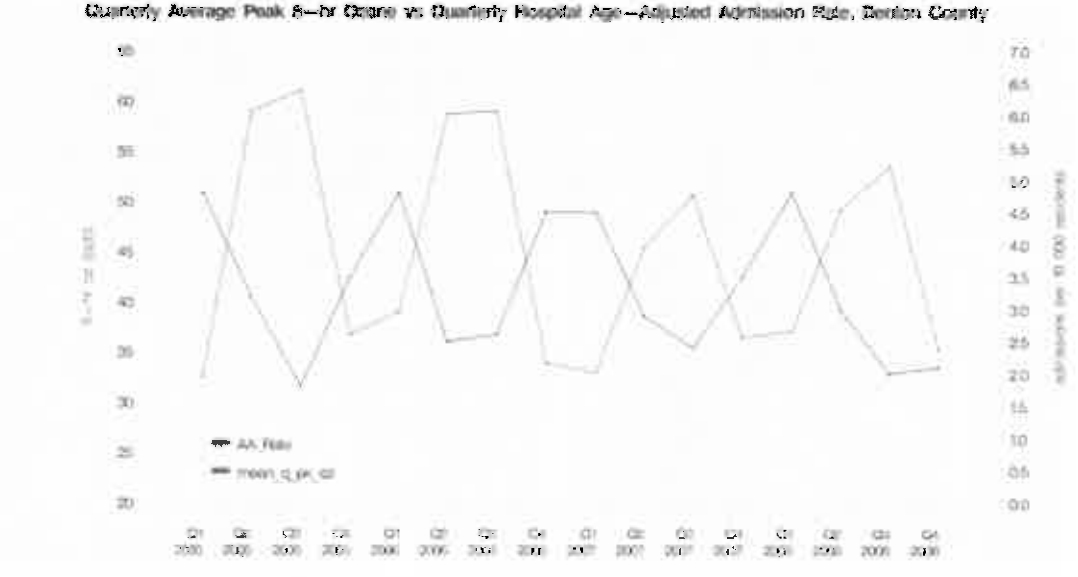
Throughout the draft HREA, EPA indicates its belief that ozone causes asthma. In fact, CASAC has repeatedly indicated that the limited evidence on new-onset asthma should not contribute greatly to the consideration of the strength of evidence for respiratory-related effects. In addition, the draft HREA states that "[i]n the case of respiratory symptoms, the evidence is most consistently supportive of the relationship between short-term ambient O_3 metrics and respiratory symptoms and asthma medication use in children with asthma…" However, it is not clear that the findings of two multi-city studies, Schildcrout *et al.* (2006) and O'Connor *et al.* (2008) have been considered. In fact, it is more accurate to say that the evidence for this endpoint is mixed.

The TCEQ analyzed data in 2000 to determine if asthma hospitalizations correlated with ozone concentrations. It was determined that there was an inverse relationship; that is, when ozone levels were high (red lines), asthma-related hospitalizations (blue lines) were low and when ozone levels were low, asthma-related hospitalizations were higher. These results suggest that ambient ozone concentrations are not the primary cause of asthma-related hospital admissions. The full results are included as an attachment. For the purposes of illustration, results for Dallas and Denton Counties are presented below.

Dallas County



Denton County



It is important to emphasize that although the causes of asthma are not fully understood, there are many factors that influence the development and exacerbation of asthma. According to the World Health Organization, one of the strongest risk factors for developing asthma is genetic predisposition. In addition, indoor allergens (dust mites, pet dander, and presence of pests such as rodents or cockroaches) together with outdoor allergens (pollen and mold), tobacco smoke, or other triggers such as cold air, extreme emotions (anger or fear) and physical exercise can all provoke symptoms in those with asthma. The Centers for Disease Control and Prevention estimates that asthma prevalence has increased over recent years from 7.3% in 2001 to 8.4% in 2010[1]. The reason for this increase is unknown, but some scientists have suggested changes in exposure to microorganisms (hygiene hypothesis) or the rise in sedentary lifestyle (affecting lung health) and obesity (which results in inflammation) may be to blame (Delgado et al. 2008).

Mortality analysis in the draft HREA is especially problematic.

EPA estimates short-term mortality impacts based on Zanobetti and Schwartz (2008) and Smith *et al.* (2009). However, the Concentration Response Functions (CRFs) vary from negative to positive for the same city, depending on which study is selected, ozone averaging time, model specifications, and ozone season. In fact, many of these estimates are indistinguishable from zero. It is not clear how these issues were considered by EPA or how the various choices of CRFs were weighed. In addition, these studies also indicate the confounding effects of co-pollutants such as PM and sulfate, which were not adequately considered by EPA because single pollutant CRFs were utilized in the core analysis.

The important information regarding the impact of co-pollutants which is presented in the appendix for Chapter 7 is not adequately communicated in the main text of the draft HREA. Namely, that for a number of cities, the sensitivity analysis indicates that upon inclusion of PM_{10} in a co-pollutant model, virtually all of the risk estimates for short-term mortality become non-significant. In addition, use of an alternate CRF from the Zanobetti and Schwartz (2008) paper results in similar findings of largely non-significant ozone-attributable mortality.

In the HREA, Figure 7-2 presents heat maps for short-term ozone-attributable mortality. It is unclear how 149 ozone-attributable deaths occur at 40-45 ppb while no deaths are due to levels >65 ppb or that there is no discernable pattern for increased/decreased risk depending on concentration. This appears to be an artifact of assuming a linear, no-threshold relationship between mortality and ozone that leads to nonsensical results.

EPA also estimates long term mortality impacts based on Jerrett *et al.* 2009. Long-term mortality was not listed on page 7-17 and 18 of the HREA under ozone-attributable effects nor is it listed as a causal endpoint in ISA. This calls into question the appropriateness of including mortality as an endpoint in the HREA. In addition, the use of the Jerrett study is concerning, as other studies of this cohort reported no associations between long-term ozone exposure and cardiopulmonary mortality that are robust to adjustment for co-pollutants (e.g., Krewski *et al.* 2000; Pope *et al.* 2002). In addition, other long-term studies of ozone-related respiratory or cardiopulmonary mortality did not report positive associations (Goodman *et al.* 2013; Dockery *et al.* 1993; Beeson *et al.* 1998; Abbey *et al.* 1999; Chen *et al.* 2005; Miller *et al.* 2007; Lipfert *et al.* 2000 for mean O_3; Lipfert *et al.* 2006; Wang *et al.* 2009; Jerrett *et al.* 2005). Moreover, it is inappropriate to

[1] http://www.cdc.gov/nchs/data/databriefs/db94.htm

combine data across cities for a national risk estimate, given the known geographic heterogeneity of these estimates (Goodman *et al.* 2013; Smith *et al.* 2009). Finally, data relating to potential confounders, e.g. smoking rates, obesity rates, diet, medication usage, etc., was collected in 1982–1983 for the ACS study but never updated. For these reasons, the national risk estimate reported by Jerrett *et al.* (2009) should not be extrapolated throughout the U.S.

The tables below summarize the available evidence addressing long-term exposure to ozone and mortality. Overall, the results are not statistically significant (i.e. the confidence intervals include 1.0), especially in models that account for co-pollutants. Only one study (highlighted in pink) Jerrett *et al.* (2009), found a statistically significant effect of long-term ozone exposure on mortality when not corrected for co-pollutant exposure. Interestingly, the effect only occurred at temperatures above 82°F. It is well known that very warm or very cold temperatures are associated with increases in mortality (Ye *et al.* 2012). Paradoxically, the increased mortality was not observed in US regions with the highest ozone concentrations (southern California) nor in areas with the highest number of respiratory deaths (the northeastern US and the industrial Midwest). The Jerrett *et al.* (2009) study used the American Cancer Society (ACS) Cohort. Other researchers have examined the same cohort and have not found an effect of ozone on mortality.

Table 1. Epidemiologic Studies of Long-Term Ozone Exposure and Mortality

Study	Cohort	n	Time Period		Seasonal or All Year	Ozone Metric	Outcome	Copollutant(s) in Model	Risk Measure	Unit of Measure	Estimate	95% Confidence Interval
			Mortality Follow-Up	Ozone Data								
[illegible] et al. (2009)	Nurses' Health Study cohort (NS)	73 [illegible]	1976-1998	1977-1988	All year	NR	All-cause	None	Rate ratio	Varies	1.01 [illegible]	NR
Jerrett et al. (2009a)	ACS CPS-II cohort of non-smokers (CA)	6,338	1982-1998	1982-1998	All year	Monthly avg	Lung cancer	None	55 (0.9)	For 22 ppb (~10th-90th)	1.19	1.03-1.38
									65 (1.0)		1.11	0.97-1.27
							NMLC		55 (0.9)		1.03	0.89-1.19
									67 (1.0)		1.05	0.92-1.21
							Cardiopulmonary		60 (0.9)		1.06	0.93-1.20
									84 (1.2)		0.93	0.83-1.05
							All natural causes		84 (1.2)		1.12	0.99-1.26
									100 (1.4)		0.99	0.89-1.09
Lipfert et al. (2006)	US veterans cohort, 50 states	50,000	1985-1988	1985-1988	All year	Summer avg; 8-h max	All-cause mortality	None	NR	Per 1000 ppb	1.19	1.00-1.39
						Resp. diseases					1.00	0.93-1.08
Stone et al. (2008)	500 nurses, 150 metropolitan areas	500,000	1993-1998	1988-1998	All year	8-h max	All other causes[a]	None	NR	N/A	0.96	0.93-1.00
					Cool season						1.00	0.93-1.08
					Warm season						1.01	0.95-1.08
					All year		All causes[a]				1.05	0.99-1.11
					Cool season						1.03	0.93-1.14
					Warm season						1.04	0.98-1.10
					All year		Cardiopulmonary[a]				1.04	0.98-1.10
					Cool season						0.98	0.89-1.08
					Warm season						0.99	0.93-1.05
					All year		Lung cancer[a]				0.99	0.85-1.16
					Cool season						0.94	0.76-1.16
					Warm season						1.01	0.87-1.17
Gent et al. (2003)	NHS, 36 metropolitan areas (CT)	3,339	1994-1998	1977-1998	All year	Monthly avg	Coronary heart disease	None	NR (+)	For 10 ppb	1.08	0.92-1.26
									NR (-)		0.97	0.81-1.16
Jerrett et al. (2005)	ACS, 96 metropolitan areas (US)	22,909	1982-1999	1980-1991	All year	Peak	All-cause	None	62	N/A	0.99	0.98-1.01
						Mean 8-h max					0.99	0.98-1.01
						Peak	IHD				0.99	0.98-1.01
						Mean 8-h max					0.99	0.98-1.01
						Peak	Cerebrovascular				0.97	0.93-0.99
						Mean 8-h max					0.99	0.98-1.01
						Peak	Lung cancer				0.99	0.98-1.01
						Mean 8-h max					0.99	0.98-1.01
Lipfert et al. (2000a)	US veterans cohort, 31 cities	70,000	1987-1991	1987-1991	All year	Peak	All-cause	None	NR	Per 5 ppb	1.03	NR
						24-hr avg		None			0.93	NR
						Peak		PM10			1.03	NR
Lipfert et al. (2000a)	US veterans cohort, 31 cities	70,000	1985-1991	1985-1991	All year	Peak	All-cause	None	NR	For 40 ppb	1.014	1.00-1.03
								PM			1.000	1.00-1.00
			1980-1991	1980-1991	All year, 24-hour max 70 ppb			None		For 50 ppb	1.018	0.99-1.04
								PM			1.000	0.99-1.01

Study	Cohort	n	Time Period of Mortality Analysis	Time Period of Ozone Data	Seasonal or All Year	Ozone Metric	Outcome	Copollutant(s) in Model	Risk Measure	Unit of Measure	Estimate	95% Confidence Interval
Jerrett et al. (2009)	ACS cohort, 96 metropolitan areas (US)	442,850	1982-2000	1977-2000; 1999-2000 (PM)	Summer only (April-Sept)	5-hr max	Allcause	None	RR	Per 10 ppb	1.002	1.00-1.01
								PM			0.999	0.98-1.00
							Cardiopulmonary	None			1.018	1.01-1.02
								PM			0.997	0.98-1.00
							Respiratory	None			1.017	1.01-1.03
								PM			1.040	1.01-1.07
							CV	None			1.014	1.01-1.02
								PM			0.995	0.97-0.99
							IHD	None			1.017	1.01-1.03
								PM			0.973	0.96-0.99
Krewski et al. (2009)	ACS cohort, 156 metropolitan areas	1.2 mil	1982-2000	1980	All year	5-hr max	Allcause	None	HR	Per 10 ppb	1.00	0.99-1.01
					Summer (April-Sept)						1.02	1.01-1.02
					All year			PM			0.99	0.98-1.00
					All year		Cardiopulmonary	None			1.02	1.00-1.03
					Summer (April-Sept)						1.03	1.00-1.04
					All year			PM			0.99	0.98-1.01
					All year		IHD	None			1.02	0.99-1.04
					Summer (April-Sept)						1.01	0.99-1.02
					All year			PM			0.98	0.98-1.02
					All year		Lung cancer	None			1.00	0.98-1.04
					Summer (April-Sept)						0.99	0.98-1.02
					All year			PM			0.97	0.93-1.03
					All year		All other causes	None			0.98	0.97-1.00
					Summer (April-Sept)						1.01	1.00-1.02
					All year			PM			1.04	0.99-1.10
Smith et al. (2009)	ACS cohort, 96 cities	352,000	1982-2000	NR	Warm season (second and third quarters)	5-hr max	Allcause	None	RR	Per 10 ppb	1.02	1.00-1.03
								Carbon and sulfate			1.00	0.99-1.01
								Carbon			1.00	0.99-1.02
				2003-2005			Cardiopulmonary	None			1.03	1.01-1.05
								Carbon and sulfate			1.02	1.01-1.04
								Carbon			1.02	1.00-1.04
Wang et al. (2009)	Cohort in Brisbane, Australia	657,955	1996-2004	1996-2004	All year	5-hr max	Cardiorespiratory	None		Per 5 ppb	1.001	0.99-1.02
								NO2 and SO2			0.999	0.99-1.01

Only for temperatures >82°F, NOT in US regions with highest ozone concentrations (Southern CA) NOR in areas with highest respiratory deaths (NE and Industrial MW).

However, since EPA used the Jerrett study as the basis for their analysis, they should use the model that best fit the data. The figure below, taken from the Jerrett study, illustrates that a threshold model fits the data better than a non-threshold model. That is, there is a safe level of exposure to ozone. However, this was not the model that was utilized by EPA.

> Figure 2 illustrates the shape of the relation between exposure to ozone and death from respiratory causes. There was limited evidence that a threshold model specification improved model fit as compared with a nonthreshold linear model (P=0.06) (Table 3S in the Supplementary Appendix).

(p. 1091-1092)

Also from Jerrett *et al.* (p. 1094):

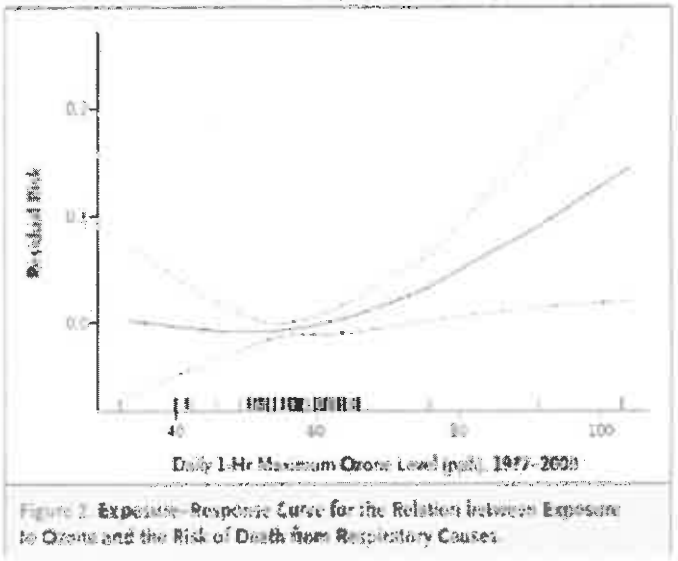
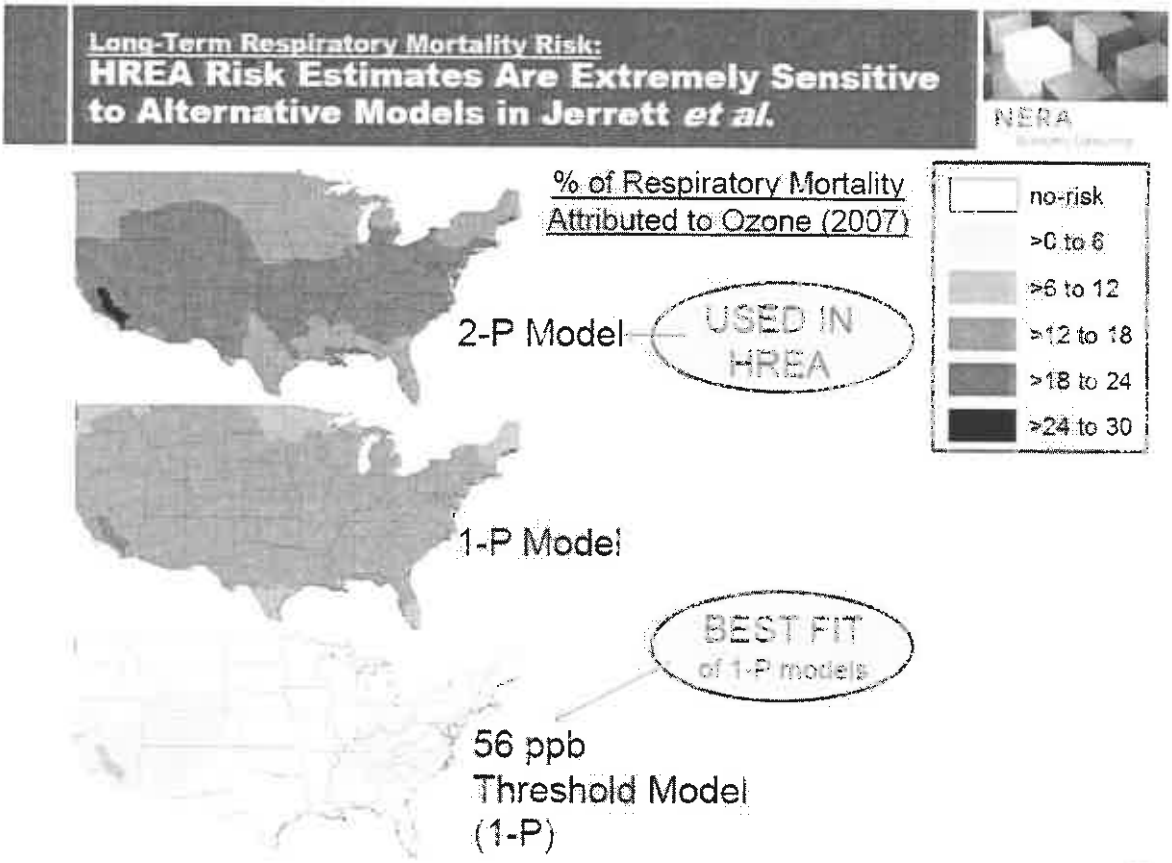
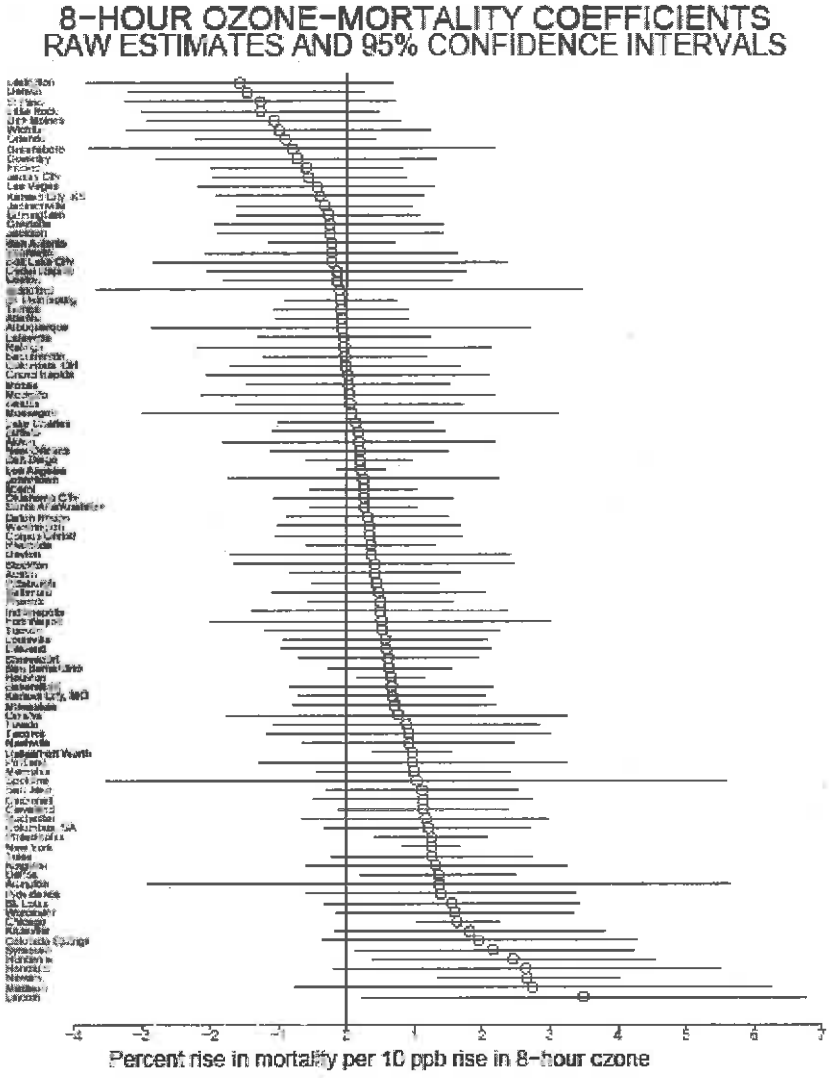


Figure 2. Exposure–Response Curve for the Relation between Exposure to Ozone and the Risk of Death from Respiratory Causes.

This is an important point because it has a tremendous impact on the calculation of risk attributed to ozone as illustrated below. Using the most scientifically-appropriate model indicates that most of the country has little to no risk from ozone, a substantially different finding than the EPA reported when using the inappropriate model.



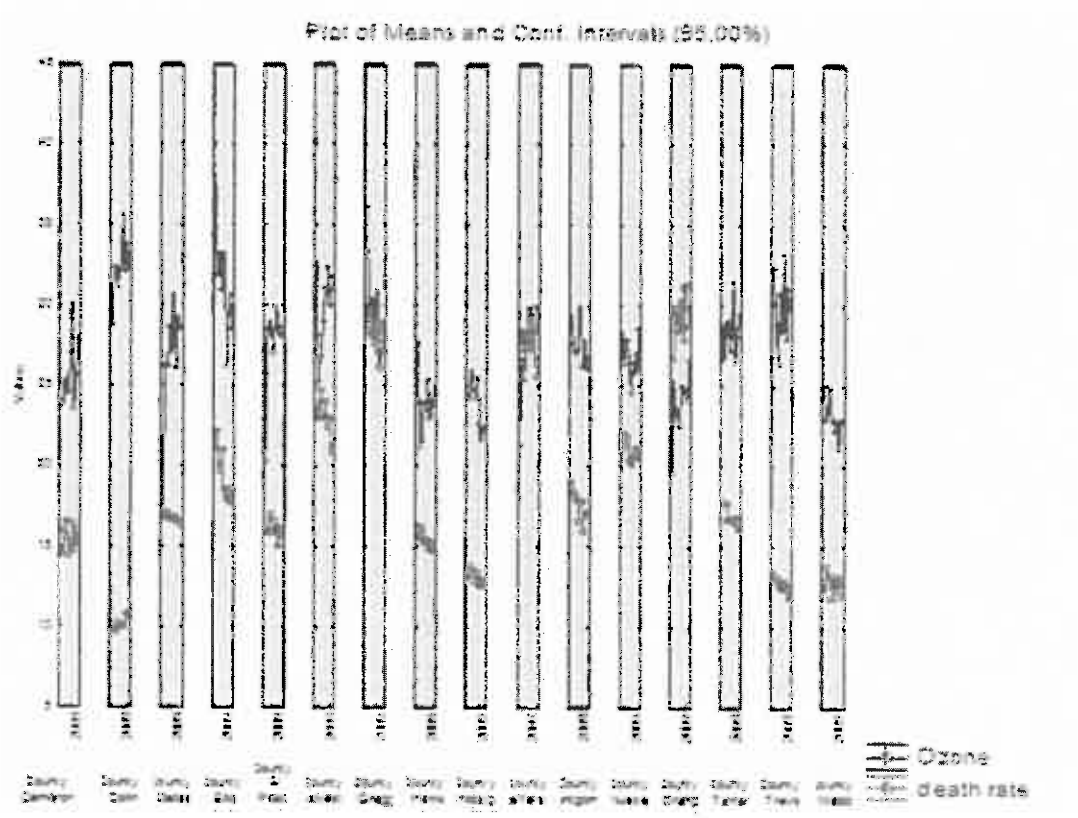
National estimates for mortality in the presence of substantial regional heterogeneity in effects estimates are especially problematic. Indeed, Smith *et al.* 2009 state "…quoting a single value as a national average is misleading if there is substantial heterogeneity." They continue "…the heterogeneity and sensitivity of ozone effect estimates to a variety of covariates leaves open the issue of whether or not ozone is causally related to mortality. Consequently the question arises whether any particular ozone-mortality effect estimate can reliably be used to predict mortality reductions that would ensue from specific ozone reductions." The figure below from Smith *et al.* 2009 illustrates the mean percent change in mortality per 10 ppb increase in 8-hour ozone concentration in 98 US cities. The vast majority show no association between ozone and mortality (cities where the horizontal bars touch the vertical, zero-effect line have no significant association). The data could also be interpreted as having a health-protective effect in some cities.



We read with interest the statement by EPA on page 7-69 of the HREA that mortality risk is generally not responsive to alternate standards. In other words, the proposed standards would not be expected to have a significant impact on mortality risk. It would then follow that EPA anticipates that there will be no appreciable benefits expected from the proposed alternative standards for this endpoint.

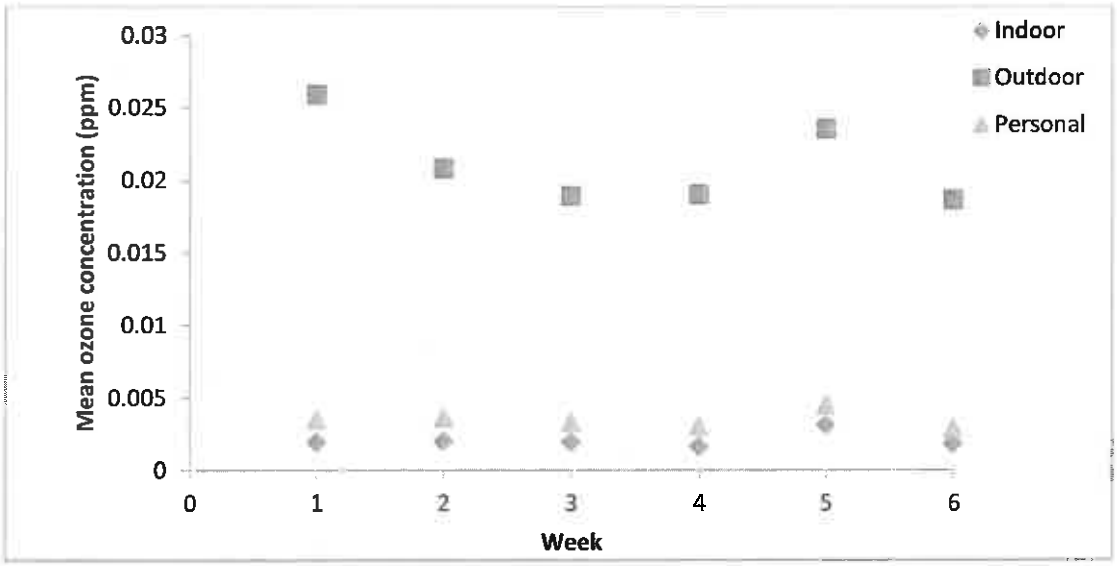
The TCEQ has worked with Louis Anthony Cox, Jr., a noted statistician and editor in chief of the journal *Risk Analysis* to examine the relationship between ozone and mortality in Texas. Between 1999 and 2010, levels of ozone in Texas varied substantially from year to year. This provides an opportunity to compare changes in ambient pollutant levels over time to changes in all-cause, cardiovascular, and respiratory mortality rates. We found that changes in historical ozone levels did not predict changes in mortality rates. Nonparametric tests also show no significant associations between yearly changes in ozone levels and corresponding changes in mortality rates. The figure below is a time series of mortality rates (deaths per million people per day) and average ozone concentrations from 2000 to 2010 by county. Note that there is no

correspondence between changes in ozone concentration (blue lines) and changes in the death rate (red lines). These findings suggest that the substantial short-term mortality benefits from reducing ozone predicted by the United States Environmental Protection Agency and others, based on statistical models of exposure-response associations elsewhere, may not hold in Texas counties.



Epidemiology studies are limited by their study design.

In EPA's discussion of the available epidemiological studies, there is inadequate discussion of personal exposure and indoor versus outdoor ozone concentrations. The graph below reproduced from Lee *et al*, 2004 illustrates results similar to numerous other studies. Because people spend much of their time indoors and indoor concentrations of ozone are extremely low, people are actually exposed to much lower ozone concentrations that those measured at outdoor monitoring stations.



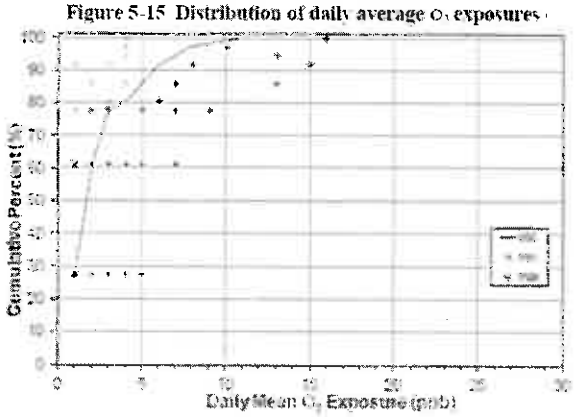
EPA should consider such differences when interpreting studies reporting associations between health effects and ambient ozone concentrations. How likely are these associations to be plausible given estimates of personal exposure? Were the *all* of the hundreds of thousands of people in the epidemiology studies outside for at least 8 hours each day immediately prior to their deaths? It is highly unlikely that these associations are plausible. In fact, in a June 5, 2006 letter to EPA from the CASAC ozone review panel, EPA's scientific advisors stated, "The Ozone Staff Paper should consider the problem of exposure measurement error in ozone mortality time-series studies. It is known that personal exposure to ozone is not reflected adequately, and sometimes not at all, by ozone concentrations measured at central monitoring sites....Therefore, it seems unlikely that the observed associations between short-term ozone concentrations and daily mortality are due solely to ozone itself."

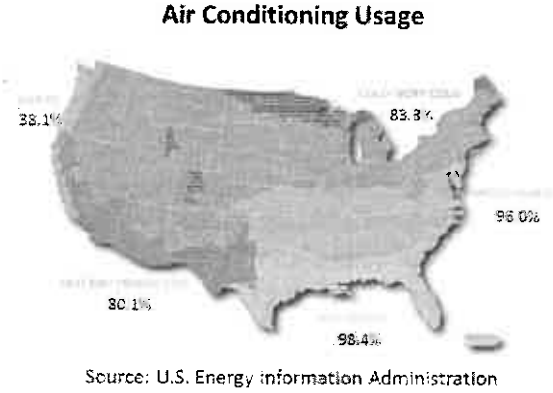
In addition, EPA introduces the topic of regional heterogeneity and states that "a national or combined analysis may not be appropriate..." in the context of discussion thresholds. However, this also calls into question the appropriateness of a one-size-fits-all standard. The observed city to city heterogeneity strongly implies such as standard would be more or less stringent than necessary, depending on location.

In conclusion, the available epidemiology studies have reported substantial heterogeneity between cities that range from positive to null or negative (i.e. higher ozone levels are correlated with reduced mortality). Therefore, a pooled nation-wide estimate is misleading and overestimates the risk of ozone. Ecological epidemiology studies are not rigorous enough to use as the basis for setting the ozone standard. EPA should use a quantitative weight-of-evidence approach that includes all available information.

Ambient concentrations are not representative of personal exposures.

EPA should explain the limitations of setting standard for ambient air based on clinical exposures when HREA states that most people spend the majority of their time indoors (see quote from CASAC above). Presumably, the patients in the epidemiology studies used by EPA to propose lowering the standard also spent much of their time indoors. Similarly, it is unclear how the results of APEX modeling in the HREA were paired with the information from the DEARS (Meng *et al.* 2012, see figure below), Xue *et al.* 2004 and Geyh *et al.* 2000 studies which indicate that daily personal exposure is well below any of the benchmarks suggested. In addition, the U.S. Energy Information Administration estimates that for many areas of the country, as much as 98.4% of the population utilizes air conditioning units, which would remove the vast majority of ambient ozone (see figure below).[2]





Source: U.S. Energy Information Administration

EPA considers outdoor workers to be an "at risk" population that may be exposed to levels of ozone reported at ambient monitors. A study by O'Neill *et al.* 2003 reported that outdoor workers in Mexico City experienced average personal ozone exposures that were 60 percent lower than ambient monitor levels. EPA also suggests that children playing outside for extended periods of time may be exposed to levels of ozone reported at ambient monitors. In a study by Lee *et al.* 2004, children in the top 25% of time spent outdoors experienced personal ozone exposures 80% lower than levels measured at ambient monitors. This difference between ambient ozone concentrations and personal exposures is key for interpreting both epidemiological studies as well as clinical exposure studies. In fact, EPA is aware that there are differences between ambient concentrations of ozone and personal exposure, but effectively ignores this difference in the HREA when deriving quantitative estimates of risk.

EPA points out in figure 5-15 of the HREA that the upper end of daily average ozone personal exposure are well less than 20 ppb, well below the current standard and the range of proposed alternate standards. EPA should consider personal exposure in setting the ozone standard, which would lead to the conclusion that the current standard is more than adequately health–protective.

[2] http://www.eia.gov/consumption/residential/reports/2009/air-conditioning.cfm

Risk is calculated below background and lowest measured levels of relevant studies.

In the draft HREA, EPA acknowledges that there is uncertainty in extrapolating health risks from ozone exposures that go beyond the ozone levels measured in the relevant epidemiology. However, EPA presents analysis on "total" risk modeled down to zero, outside of the range of available data. This is problematic because there is no way to determine the uncertainty surrounding the risk estimates for the alternative standards under consideration.

In reviewing the studies cited by EPA in the HREA, associations between ozone and selected endpoints generally became weaker and not significant at lower ozone levels. EPA did not incorporate these findings in its risk assessment. Instead, risks were extrapolated below the Lowest Measured Levels (LMLs) of the selected studies and to zero ozone, even though the data from the underlying studies did not report effects at low levels of ozone.

Perhaps more importantly, in assigning risk below background levels of ozone, EPA is suggesting risk below levels that can be potentially modified by implementation of the ozone NAAQS, as pointed out by CASAC in its review of the first draft HREA. In fact, one member of CASAC stated "The C-R function which goes down to zero makes little sense. First of all, such levels are never obtained… Secondly, this zone has little value since it cannot be influenced by the regulatory process." This commenter continues "…we should have a vision of what levels/cut offs are scientifically sound and contribute to standard setting in a practical way." A second commenter added "[g]iven the background levels of O_3 that cannot be controlled by U.S. regulatory actions, this reviewer endorses applying the C-R function down to the LML and does not support obtaining risk estimated down to zero."

Given the uncertainty surrounding risks calculated at low levels of ozone, EPA should assess risk above background ozone levels, as these are the levels that can potentially be controlled by regulation.

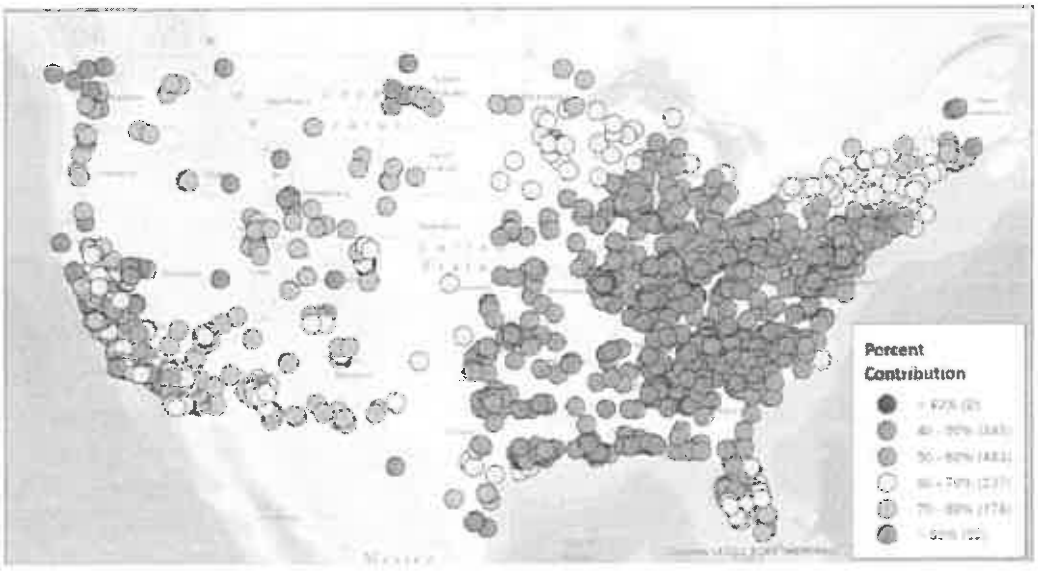


Figure 2-12. Map of apportionment-based U.S. background percent contribution to seasonal mean O_3 based on 2007 CAMx source apportionment modeling.

There is substantial evidence for confounding by co-pollutants.

The core analysis presented in the draft HREA includes estimates for single pollutant models. However, EPA noted in the first draft HREA that confounding by co-pollutants reduces the effect estimates for ozone. Therefore EPA should acknowledge that risk estimates may well be overestimated by not using multi-pollutant models. In fact, CASAC also commented on this point: "[t]o this reviewer, no results should be presented that have not taken into account $PM_{2.5}$ at a minimum." This topic is especially troubling as the additional analysis presented in Appendix 7 demonstrates that upon inclusion of PM_{10} in a co-pollutant model, virtually all of the risk estimates for short-term mortality become non-significant.

EPA should utilize multi-pollutant models that account for the confounding effects of co-pollutants and better capture the potential contribution of ozone to health effects.

The rationale for lower ozone standard is inadequate.

The draft HREA presents hypothetical health effects that are based on one or two 8-hour theoretical exposures above the various benchmarks. However, the ozone standard is based on the 4th highest 8-hour exposure averaged over 3 years. It is not clear how this analysis supports a lower standard that would not necessarily capture a single exposure over a given benchmark.

The draft HREA presents modeling results for Houston stating that "seasonal average values …remained nearly constant relative to the existing standard when air quality were [sic] further adjusted to meet the 65 ppb standard." This observation does not support lowering the NAAQS.

The mortality estimates for alternative standards presented by EPA generate surprising, nonsensical results. Net mortality was estimated to *increase* in cities including Houston under alternative standards. In addition, it seems highly unlikely that the majority of the risk calculated for ozone (~75%) is attributable to low ozone concentrations (<60 ppb, which is the lower bound for the proposed alternative standards). Furthermore, for Houston <1% of mortality risk is estimated for ozone concentrations >60 ppb, based on Figures 7-2 and 7-3. We fail to see how cities such as Houston would be expected to benefit from the alternative standards proposed because EPA estimates *increased* mortality from lowering the standard.

Based on Table 5-7 of the HREA it appears that the only significant potential exposures would be to 60 ppb ozone. According to the scientific literature, at this concentration one would expect only mild, reversible, transient effects on lung function that are of unclear clinical importance. Furthermore, based on the confidence intervals presented in this table, no significant exposure to 70 or 80 ppb would be expected even if the current standard were to be retained. Therefore, it is not clear how this information supports a more stringent NAAQS.

Finally, the last line of the HREA states: "[m]ortality from short-term and long-term O_3 exposures and respiratory hospitalization risk is not greatly affected by meeting lower standards…" This observation does not support the necessity of a lower standard. EPA's own modeling shows either adverse or little to no public health benefit from lowering the current standard.

EPA does not present a clear rationale for the necessity of a lower standard. For instance, the evidence presented by for respiratory endpoints EPA appears to cast doubt on the lower end of the proposed range of alternative standards. In addition, EPA indicates "…a *mostly consistent*

positive association between O_3 exposure and respiratory-related hospital admissions and ED visits…" "Mostly consistent" is not strong enough evidence for using these endpoints for setting a lower NAAQS.

In addition, the three observations on page 3-112 of the HREA are based on mortality over the full range of ozone concentrations. Based on figure 3-16, the choice of a 60—70 ppb standard will not appreciably change any of these key observations. Moreover, the final line of the HREA states "[m]ortality from short-term and long-term O_3 exposures and respiratory hospitalization risk is not greatly affected by meeting lower standards…" We agree with EPA that the proposed alternate standards will not have an impact on respiratory hospitalization risk, therefore there is no scientific reason to lower the existing standard.

Finally, on page 3-115 of the PA and elsewhere in discussion of uncertainty related to effects at low concentrations of ozone EPA makes contradictory statements. EPA calculates increases in theoretical mortality resulting from alternative standards leading to potentially substantial disbenefits. However, EPA argues that the decreases in health effects estimated for higher ozone concentrations are real whereas the increases in those same health effects at lower concentrations are uncertain. It can't be both ways. This illustrates the impact of choosing a linear model and calculating risks below both the LMLs of available studies as well as background O_3. This also highlights the tenuous connection between the ozone and mortality, especially at concentrations below the current standard.

EPA's own modeling shows either adverse or little to no public health benefit from lowering the current standard, therefore EPA should retain the existing standard.

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Appendix B
Ohio University Letter to Director Butler on Proposed Rule for the Ozone NAAQS

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March 6, 2015

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RE: 2014 US Environmental Protection Agency (EPA) Proposed Rule for the Ozone National Ambient Air Quality Standard.

Dear Director Butler,

This letter is intended to provide additional comments to the proposed U.S. EPA revision to the ozone standard, with the intent that this comment will be included in the overall response by Ohio EPA.

The proposed rule making would lower the primary standard within the range of 65-70 ppb while retaining the averaging time and statistical evaluations of the current standard to determine non-compliance. Specifically the following comment submitted by the Center for Air Quality at Ohio University is centered on the predictive power of compliance monitors on personal exposures.

The Center for Air Quality has been involved with air quality issues within the State over the last twenty years. The Center for Air Quality, as part of the Institute for Sustainability Energy and the Environment (ISEE) and the Consortium for Energy Economics and the Environment (CE3) at Ohio University, has been addressing energy and environmental issues in Ohio and the region. This includes research on several key air quality and energy issues including comprehensive modeling and monitoring projects on ozone, and particulate matter (PM). In addition, the Center for Air Quality has conducted daily forecasts and hourly "real time" ozone and particulate matter mapping for a majority of the metropolitan centers within the State of Ohio. The mapping involved identifying and displaying the temporal and spatial characteristics of ozone and PM concentrations.

Our comment on the proposed ozone standard revision involves the expansion of the ozone season and the predictive power of compliance monitors on personal exposures. The proposed standard, which will reduce the time weighted average ozone concentration in the range of 65-70 ppb, will expand the ozone season from very early spring until late fall for the mid to northern latitudes. At this lower proposed level an exceedance will occur under meteorological conditions that will amplify spatial variations in the ozone concentrations. This could potentially result in narrow, elevated ozone "bands" during early spring and the late fall months that would not be representative of the ozone concentration in the local air shed.

There have been numerous studies calling into question the predictive power of compliance monitors on personal exposure to ozone. Often cited in peer-reviewed publications are the lack of correlation between

indoor and outdoor ozone concentrations and the large spatial variations of outdoor concentrations. Factoring in the expanded ozone season will only further add to this lack of correlation.

This issue of utilizing compliance monitors to predict human exposure has also been a concern for US EPA. This concern was recently highlighted in the June 2014 request for proposal under the National Center for Environmental Research Science to Achieve Results (STAR) program titled *Air Pollution Monitoring for Communities*, which was addressing the need for an expanded monitoring network involving personal monitors.

Finally, in Ohio a typical ozone exceedance occurs during elevated temperatures and or stagnant/ stable atmospheric conditions. During these warmer months an exceedance in the 65-70 ppb range will be highly influenced by background ozone concentrations. In this scenario transported ozone and biogenic emissions will play a significant role in the exceedance. Under this condition identifying effective local control strategies for reducing ozone precursors will be extremely difficult. In addition, the potential expansion of the monitoring network will be necessary to capture the influence of background ozone concentrations on non-attainment areas.

In conclusion, an evaluation of the current national ambient air-monitoring network should be conducted to determine if the present system could support the proposed lowering of the ozone standard before the final rule is issued. US EPA should fund the States for further study of these issues. This will ensure that cost effective control strategies can be constructed.

Thank you for the opportunity to include the Center for Air Quality's comments within the package submitted to U.S. EPA from the State of Ohio.

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Appendix C-1
Outdoor/Indoor Personal Ozone Exposures of Children in Nashville, Tennessee

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http://www.tandfonline.com/loi/uawm20

Outdoor/Indoor/Personal Ozone Exposures of Children in Nashville, Tennessee

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Published online: 21 Feb 2012.

To cite this article: Kiyoung Lee , William J. Parkhurst , Jianping Xue , Halûk Özkaynak , Donna Neuberg & John D. Spengler (2004) Outdoor/Indoor/Personal Ozone Exposures of Children in Nashville, Tennessee, Journal of the Air & Waste Management Association, 54:3, 352-359, DOI: 10.1080/10473289.2004.10470904

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TECHNICAL PAPER

ISSN 1047-3289 J. Air & Waste Manage. Assoc. 54:352–359
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Outdoor/Indoor/Personal Ozone Exposures of Children in Nashville, Tennessee

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ABSTRACT

An ozone (O_3) exposure study was conducted in Nashville, TN, using passive O_3 samplers to measure six weekly outdoor, indoor, and personal O_3 exposure estimates for a group of 10- to 12-yr-old elementary school children. Thirty-six children from two Nashville area communities (Inglewood and Hendersonville) participated in the O_3 sampling program, and 99 children provided additional time-activity information by telephone interview. By design, this study coincided with the 1994 Nashville/Middle Tennessee Ozone Study conducted by the Southern Oxidants Study, which provided enhanced continuous ambient O_3 monitoring across the Nashville area. Passive sampling estimated weekly average outdoor O_3 concentrations from 0.011 to 0.030 ppm in the urban Inglewood community and from 0.015 to 0.042 ppm in suburban Hendersonville. The maximum 1- and 8-hr ambient concentrations encountered at the Hendersonville continuous monitor exceeded the levels of the 1- and 8-hr metrics for the O_3 National Ambient Air Quality Standard. Weekly average personal O_3 exposures ranged from 0.0013 to 0.0064 ppm (7–31% of outdoor levels). Personal O_3 exposures reflected the proportional amount of time spent in indoor and outdoor environments. Air-conditioned homes displayed very low indoor O_3 concentrations, and homes using open windows and fans for ventilation displayed much higher concentrations.

IMPLICATIONS

This study demonstrates the usefulness of passive O_3 sampling technology in measuring long-term outdoor/indoor/personal exposures. The test subjects did well in following simple directions concerning accurate exposure assessment and in keeping time-activity diaries. Personal O_3 exposure, in between the extremes of higher outdoor and lower indoor exposures, is a function of time spent outdoors. Clearly, those children spending more time outdoors are subject to higher O_3 exposures than are their more housebound peers. Continuous State and Local Air Monitoring System O_3 monitoring results substantially overestimate weeklong indoor and personal O_3 exposure. Centrally air-conditioned indoor environments confer a substantial degree of protection from ambient O_3 levels.

INTRODUCTION

Ozone (O_3) is a secondary air pollutant, produced photochemically from complex mixtures of nitrogen oxides (NO_x) and reactive volatile organic compounds (VOCs). The short-term reversible effects of O_3 on respiratory symptoms and pulmonary function have been well established by toxicological and field studies of both adults and children.[1–4] In a prospective between-community cohort study, Detels and coworkers[5,6] found decreased levels of lung function and steeper declines in lung function among residents of communities more heavily exposed to oxidant pollution. Significantly lower lung function was observed for people living in areas where annual average outdoor O_3 concentrations exceeded 0.04 ppm.[7] Other studies suggest that asthma can be exacerbated by O_3 exposure. Some have suggested that repeated exposure to photochemical smog events can cause asthma, though it is not clear which pollutant or combination of pollutants is responsible.[8] Most studies of chronic health effects are determined by regression analysis of various health endpoints based on ambient monitoring data collected from fixed monitoring sites. Imprecise exposure characterization adds to considerable uncertainty in study results.

One significant limitation of previous O_3 exposure investigations has been the lack of personal and indoor O_3 monitoring. Existing continuous O_3 monitors are bulky, expensive, and impractical for wide-scale personal and indoor exposure monitoring studies. With the

development of passive O_3 samplers,[9–12] however, researchers have demonstrated that personal and indoor O_3 monitoring is feasible. Simultaneous measurements using the Harvard O_3 passive sampler and conventional continuous analyzers show good agreement in outdoor, indoor, and personal exposure measurements.[13,14] These O_3 passive samplers were used to monitor personal O_3 exposures of 23 elementary school children, as well as the indoor and outdoor concentrations in their homes.[15] The samplers were used to measure chronic O_3 exposures of 200 children by measuring ambient, outdoor, and personal exposures once a month.[16]

This study measures weekly outdoor, indoor, and personal O_3 exposure estimates during six weeklong periods in the summer of 1994 for two communities in the Nashville, TN, metropolitan area. This study coincided with the 1994 Nashville/Middle Tennessee Ozone Study conducted by the Southern Oxidants Study, which provided enhanced continuous ambient O_3 monitoring across the Nashville area. The study also established the ability of 10- to 12-yr-old children to maintain a demanding time/activity protocol for O_3 exposure assessment. The O_3 exposure estimates were used further to determine if systematic differences among children exist as a result of differing housing characteristics, residential locations, or activity patterns.

EXPERIMENTAL METHODS

Study participants were recruited from elementary schools in the Inglewood and Hendersonville areas of Nashville. Inglewood is an urban community within the city limits of Nashville, 8 km northeast of the city's center, and Hendersonville is a suburban town, 22 km northeast of Nashville. In the summer, both Inglewood and Hendersonville are predominantly on the downwind side of the Nashville metropolitan area. The schools and their neighborhoods were selected, in part, because of their proximity to nearby continuous ambient O_3 monitors. From the two schools, 326 children were asked to complete a preliminary questionnaire. Fifty children responded affirmatively to the questions regarding their presence during the six weekly monitoring periods and had parental permission to participate the study. After initial contact by our staff, 36 of these children agreed to participate. During the study, three children dropped out (two children completed 3 weeks and one child completed 4 weeks) and 33 children completed the six weeklong measurements. The measurements were completed during the school's summer vacation in June and July of 1994.

Continuous O_3 measurements at the ambient monitoring stations were made with Federal Reference Method (FRM) O_3 monitors operated by the Nashville/Davidson County Metropolitan Health Department (Inglewood) and the Tennessee Department of Environment and Conservation (Hendersonville). Both samplers operate according to State and Local Air Monitoring Station (SLAMS) siting and quality assurance guidelines, and their hourly data is submitted to the Aerometric Information and Retrieval System (AIRS). The continuous data used for this analysis were retrieved from AIRS.

Passive outdoor/indoor/personal O_3 measurements were made using Harvard passive O_3 samplers (Ogawa and Co. USA, Inc.).[12] The sampler consists of a Teflon barrel containing two glass-fiber filters coated with nitrite. The sampler estimates O_3 using the oxidation reaction of nitrite by O_3 to form nitrate (NO_3^-). The amount of accumulated NO_3^- is determined by ion chromatography. Time-weighted average O_3 concentrations are estimated by the amount of NO_3^- and effective collection rate. Collection rates were 21.6 cm^3/min for outdoor sampling, 21.4 cm^3/min for indoor sampling (with forced air movement), and 14.8 cm^3/min for personal sampling.[13,17,18] Field blanks were deployed with field samplers every week, and they were matched at 10%. The field blank values were subtracted from the O_3 measurement.

Each participating child and family had a set of personal/indoor/outdoor passive O_3 samplers. The tripod-mounted, weather-capped, outdoor sampler was placed ~2 m off the ground and located in a secure, open area away from vents, driveways, and overhanging trees near the participant's homes. The indoor sampler was placed on a small sampling rack with an integral small fan to provide a constant face velocity of 0.25 m/sec. The fan rack was placed on bookshelves or tabletops in the homes of participants. The child was instructed to wear the passive sampler pinned to the left top front side of their clothes during the active (nonsleep) portion of the day and to place the sampler next to their bed or on a dresser at night. They also were instructed not to get the sampler wet. The children also recorded daily activities, from 8:00 a.m. to 9:00 p.m., in diaries. Information about house characteristics, activity patterns, and families of the 36 children was collected via a questionnaire before the monitoring.

Ninety-nine children were recruited from the same elementary schools to provide additional time/activity information during the summer via telephone survey. This activity information was collected for a sample of 15 nonconsecutive days. The children were informed of the interview ahead of time so that they could record their activities. The activity information was not collected retrospectively if the child was not reachable on that day. A total of 62 children completed the

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telephone interview at least eight times. These activity patterns were compared with those of the monitored children.

Outdoor, indoor, and personal O_3 concentrations along with children's activity patterns and household characteristics were described using Student's t test and one-way analysis of variance. Because there were repetitive measurements of the same child, a mixed model with time as a covariate was used to include random effects and adjust variability accordingly. The analysis determined time trends (temporal) and the individuals in which they were measured (spatial). The data were used to develop a model to predict personal O_3 exposures using a least-squares regression model.

RESULTS AND DISCUSSION

Of the 36 children participating in personal O_3 exposure monitoring, 18 were female, 28 were Caucasian, and eight were African-American. Fourteen were 10 years old, 18 were 11 years old, and four were 12 years old. Twenty-eight lived in detached single-family dwellings. Twenty-one homes had central air conditioning, 11 homes had window air-conditioning units, and one home had no air conditioning. Window fans were "often" used in eight homes, "occasionally" used in six homes, and "not" used in 21 homes. Windows were "frequently" opened in five homes, "occasionally" opened in 21 homes, and "not" opened in nine homes.

Three children dropped out and 33 children completed all six weekly measurements. Among the participating children, 96.5% of children-week personal exposures were measured. A total of 932 passive samplers were used for indoor, outdoor, and personal sampling. Only eight samples were voided because of being "lost, wet, or broken." Seven of these eight were personal samplers. One outdoor sample was lost when the supporting tripod fell during a thunderstorm. No child had more than one occurrence of voided personal sampling. Based on this experience, it is believed the passive sampler can be used effectively to measure chronic O_3 exposure for children.

Duplicate indoor measurements were used to determine the precision of the O_3 passive sampler. When both measurements were below the limit of detection, the pair was not included in the analysis. One hundred and nine duplicate samples were collected; 42 pairs had at least one measurement above the limit of detection. A relative precision of 18.5% was estimated as the standard deviation of the absolute difference between each pair (0.0011 ppm) divided by the mean indoor concentration (0.0061 ppm). The precision was slightly higher than 12% for indoor exposure measurements reported elsewhere.[16] The difference was mainly caused by low indoor levels in this study.

Activity diaries were collected during the O_3 sampling period. The children were asked to record their activities from 8:00 a.m. to 9:00 p.m. When the diary was missing data for less than 3 hr, the time missed at each location was adjusted proportionally to the data available for that day. For the documented times, the children averaged 2.8 hr outdoors, 9.3 hr indoors, and 0.9 hr "in transit."

Weekly outdoor average O_3 concentrations from the continuous monitor and the highest and lowest outdoor passive monitors in Inglewood and Hendersonville are shown in Figure 1. On average, weekly O_3 concentrations were 65% higher for Hendersonville compared with Inglewood (0.033 vs. 0.02 ppm). The 0.12 ppm level of the 1-hr National Ambient Air Quality Standards (NAAQS) for O_3 was exceeded (0.127 ppm) during week 1 in Hendersonville, and the 0.08 ppm level of the 8-hr O_3 NAAQS was exceeded four times (0.101, 0.105, 0.086, and 0.085 ppm) during weeks 1, 2, 4, and 6, respectively, in Hendersonville. Neither standard level was exceeded at the Inglewood monitor. The average ambient O_3 levels encountered during this study are typical of levels measured during the Middle Tennessee Ozone Study.[19]

The ambient O_3 levels in this study were comparable with historical data showing average O_3 levels of ~0.04 ppm during the O_3 season from April to September. Average O_3 NAAQS was exceeded in Hendersonville. Adults exercising outdoors (O_3 levels 0.021–0.074 ppm) and farm workers (O_3 levels 0.013–0.084 ppm) can experience decreased lung function at the O_3 levels observed in this study.[20,21] These ambient O_3 levels have been associated with increased emergency room visits and hospital admissions, including aggravation of

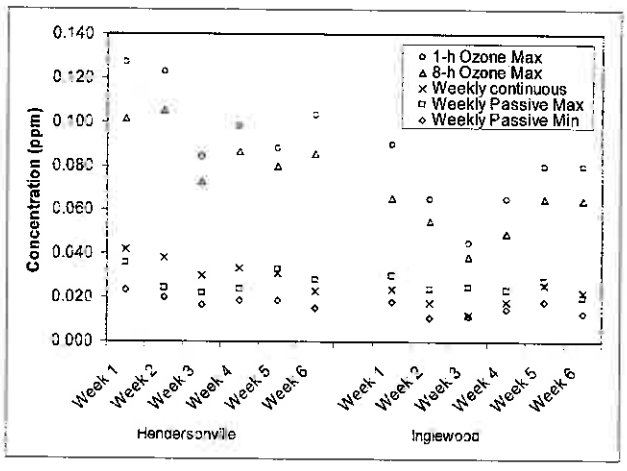

Figure 1. Ambient O_3 concentrations in Hendersonville and Inglewood.

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asthma.[22,23] However, other studies did not replicate these findings.[24]

Weekly exposure estimates from the continuous monitors correlated well with passive samplers placed outside of the subjects' homes. There is particularly good agreement between the continuous Inglewood monitoring results and the average Inglewood outdoor passive sampler results. The same cannot be said, however, for Hendersonville, where the continuous monitoring results were from 8 to 67% higher than the average outdoor passive O_3 levels. While the area immediately around the Inglewood monitor is similar to the residential area studied, the Hendersonville monitor is relatively more isolated from its nearby residential area. The Hendersonville residential area, like all residential areas, contains many on-road and off-road NO_x and VOC emissions sources (cars, trucks, and lawnmowers), which expose the passive samplers to a far more dynamic chemical regime than that found at the Hendersonville continuous monitor. The Hendersonville monitoring station was located in a limited access area next to a city park on the north shore of Old Hickory Lake. The net effect of the proximity of these "fresh" emissions is to lower O_3 concentrations through chemical titration.

Descriptive statistics for weekly outdoor, indoor, and personal passive O_3 exposure estimates are shown in Table 1. All of the outdoor passive samples estimated concentrations above the minimum detection limit (MDL) of 0.0012 ppm, although 64% of the indoor passive samples and 40% of the personal samples were below the MDL. For the samplers where the concentrations were below the limit of detection, the O_3 concentration was assigned as half of the lower detection limit (i.e., 0.0006 ppm) for statistical analyses. Weekly outdoor passive O_3 measurements in Hendersonville averaged 6–24% higher than those for Inglewood, and this difference was significant for five of the six weeks ($p < 0.01$). The regional difference in ambient O_3 exposure between Inglewood and Hendersonville demonstrated in this study is consistent with the results of a 3-yr ambient O_3 monitoring study.[19] Weekly average indoor O_3 concentrations ranged from 3 to 15% of outdoor O_3 concentrations, and average personal O_3 concentrations ranged from 7 to 31% of outdoor O_3 concentrations.

More than 60% of indoor samplers were below the MDL. Because there are very few indoor sources of O_3, and because outdoor O_3 is effectively removed by air conditioning, indoor O_3 levels are often low. In this study, the average ratio of indoor/outdoor O_3 was 0.1 with a standard deviation of 0.18. The ratio is less than ratios of 0.26 and 0.40 reported in California and Hong Kong, respectively.[25,26] Forty percent of personal samples were below the MDL. The percentage of personal samples below the MDL was lower than that for indoor samples. This is reasonable given that personal exposure depends on both indoor and outdoor levels.

When housing characteristics and indoor O_3 levels were assessed, four variables were significantly associated

Table 1. Descriptive statistics for indoor, outdoor, and personal O_3 concentration (ppm).

Week	Exposure	No.	Mean	Standard Deviation	Min	Median	Max	No. of Samples below Detection Limit (%)
1	indoor	35	0.0019	0.0035	0.0006	0.0006	0.017	30 (86)
	outdoor	35	0.0259	0.0031	0.018	0.0254	0.0356	0
	personal	34	0.0035	0.0032	0.0006	0.0023	0.013	11 (32)
2	indoor	35	0.002	0.0035	0.0006	0.0006	0.0179	24 (69)
	outdoor	35	0.0208	0.0029	0.0112	0.0209	0.0245	0
	personal	35	0.0036	0.0038	0.0006	0.0022	0.0167	13 (37)
3	indoor	35	0.0019	0.0029	0.0006	0.0006	0.0129	23 (66)
	outdoor	34	0.0189	0.0023	0.0117	0.0185	0.0247	0
	personal	32	0.0033	0.0033	0.0006	0.0022	0.0128	13 (41)
4	indoor	33	0.0016	0.0025	0.0006	0.0006	0.0133	26 (79)
	outdoor	32	0.019	0.0023	0.0147	0.0185	0.0238	0
	personal	33	0.003	0.0056	0.0006	0.0006	0.0254	22 (67)
5	indoor	32	0.0031	0.0034	0.0006	0.0018	0.0147	4 (13)
	outdoor	32	0.0235	0.0031	0.018	0.0226	0.0329	0
	personal	31	0.0045	0.0037	0.0006	0.0026	0.013	6 (19)
6	indoor	32	0.0018	0.0027	0.0006	0.0006	0.011	22 (69)
	outdoor	32	0.0186	0.0032	0.0127	0.0184	0.0284	0
	personal	33	0.0029	0.0028	0.0006	0.002	0.0103	15 (45)

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with indoor O_3 levels: carpeting, air conditioning, window fans, and window opening ($p < 0.05$). The effect of carpeting is inconclusive because there were only two houses without carpet in this study. Indoor O_3 concentrations were significantly lower for houses with central air conditioning, houses that did not use window fans, and houses that did not open windows. When the one nonair-conditioned house was excluded, the three variables were not associated with each other. These three variables seem to be good predictors of lower indoor O_3 concentrations.

No housing characteristic was significantly associated with personal O_3 exposure over all six weeks. The presence of a pet was significantly associated with higher personal O_3 exposures for four of the six weeks. Subsequent analysis between variables and activity patterns indicated that children with pets tended to spend more time outdoors than children without pets. The presence of a pet appears to be a surrogate for outdoor activity, which results in higher personal O_3 exposure. When repeated measures analysis using the mixed model was used to explore models for personal exposure, the final model included three variables, 3 or more hours of time spent outdoors, the presence of a pet in the home, and the use of a window fan. None of the interaction terms between these variables were found to be significant. The 6-week average personal O_3 exposures were classified by the selected variables, as shown in Table 2.

The importance of time spent outdoors and indoors relative to personal exposure was examined. When personal O_3 exposures were stratified into three groups based on outdoor time (<25th percentile, 25–75, >75th percentile), the exposures of children spending the least amount of time outdoors were significantly lower than those of children spending the greatest amount of time outdoors (Table 3). From this, it can again be concluded that the time spent outdoors is a strong indicator of personal O_3 exposures. McConnell et al.[27] reported that the incidence of new asthma was associated with heavy outdoor exercise in communities with high ambient O_3 concentrations. This finding indirectly supports the importance of outdoor time in personal O_3 exposure. However, it was found that no housing characteristics were associated with personal exposure. This suggests that indoor levels are less important for personal exposure. The lack of correlation between housing characteristics and personal exposure may not be generalized, because such an association may be caused by very low indoor levels in this particular study.

Table 3. Classification of O_3 concentrations (ppm) into three groups by outdoor time from activity diary without one child living in non-air-conditioned house.

Outdoor Time	Personal	Indoor	Outdoor
<25th percentile	0.0015 ± 0.00074	0.00108 ± 0.00136	0.02141 ± 0.00194
25–75	0.002 ± 0.0016	0.00105 ± 0.00103	0.02122 ± 0.00206
>75th percentile	0.0042 ± 0.0026	0.00349 ± 0.00478	0.02106 ± 0.0023

Correlations between personal exposure and percentages of weekly indoor, outdoor, and transit times were determined. Correlations between personal/indoor ratios and personal/outdoor ratios also were made with the percentages of time spent indoors, outdoors, and in transit. The activity data were included only for times when the children wore the sampler. The Pearson correlations are shown in Table 4. Personal O_3 exposure was inversely associated with time spent indoors and positively associated with time spent outdoors. Transit time was not significantly associated with personal exposure. The relationship between outdoor time-of-day and personal exposure was similar to a daily profile of ambient O_3 concentration, as shown in Figure 2. Outdoor O_3 concentrations usually have a daily profile in which O_3 concentrations are elevated in the afternoon and low in the early morning.[19] The diurnal profile suggests that the personal O_3 exposures were affected not only by the total amount of time spent outdoors but also the time of day the child was outdoors.

Table 2. Six-week average personal O_3 exposure by the selected variables from repeated measures analysis using the mixed model.

Variables	Classification	Personal Exposure (ppm) ± Standard Deviation
Pet	No	1.46 ± 0.71
	Yes	3.93 ± 2.7
Window fan	Never	1.31 ± 1.96
	Occasionally	2.11 ± 1.55
	Often	4.04 ± 3.66
Outdoor time yesterday	<3 hr	1.32 ± 1.27
	>3 hr	2.63 ± 2.39

Table 4. Correlation between personal exposure and activity.

	Personal Exposure	Personal/Indoor	Personal/Outdoor
Indoor time	−0.17[a]	−0.1[a]	−0.18[a]
Outdoor time	0.19[a]	0.1[a]	0.2[a]
Transit time	−0.01	0.01	−0.02

[a] $p < 0.01$

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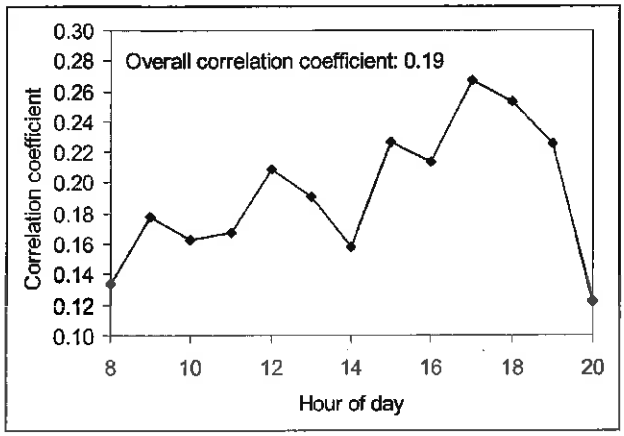

Figure 2. Correlation between personal exposure and hour of day when outdoors.

Activity patterns were collected from two groups, the 36 children participating in the O_3 monitoring study and the 99 children taking part in the telephone interview. Of the group of 99 children, activity patterns of 62 children who completed the telephone interview at least eight times were included for the analysis. Of these 62 children, 37 were female. Sixty children lived in houses with air conditioning (25 had central air conditioning and 35 had window air conditioning). The average time spent outdoors for these children was 4.1 of 13 hr, substantially higher than the average of 2.8 hr outdoors obtained from those children participating in the O_3 monitoring study.

The telephone survey group averaged 1.3 hr more time outdoors, between 8:00 a.m. and 9:00 p.m., than the group monitored for personal O_3. Figure 3 shows that ~35% of children in the monitoring group spent all 13 hr indoors, while this was true for only 5% of children in the telephone survey group. Despite these apparent differences in reported time spent outdoors, a direct comparison cannot be made between the groups because the information collection instruments were designed and used differently. It is possible that participants in the O_3 study may actually have behaved differently with respect to time spent outdoors than did other children in the Nashville area. However, it is not clear whether these results suggest a real difference in the behavior of the O_3 sampled children caused by selection criteria or a bias introduced by differences in the assessment methods. Further investigation is needed on the impact of strict participation requirements on exposure profile.

Methods of predicting personal O_3 exposures were examined using the data on microenvironmental O_3 concentrations and daily time-activity diaries. Because this study monitored only two microenvironments, indoors and outdoors at-home, two basic models were tested using an ordinary least-squares regression model:

$$C_p = \beta_1 C_i + \beta_2 C_o + \varepsilon \quad (1)$$

and

$$C_p = \beta_1 f_i C_i + \beta_2 f_o C_o + \varepsilon \quad (2)$$

where C_p, C_i, and C_o are the personal, indoor, and outdoor O_3 concentrations, respectively; f_i and f_o are the fraction of time spent indoors and outdoors, respectively; and ϵ is the error term.

The model using unweighted O_3 concentrations (eq 1) was

$$C_p = -0.61 + 0.40\, C_i + 0.11\, C_o + \varepsilon \quad (3)$$

The model using weighted microenvironmental O_3 concentrations (eq 2) was

$$C_p = 1.20 + 0.46\, f_i C_i + 0.34\, f_2 C_o + \varepsilon \quad (4)$$

In both models, the coefficients of indoor and outdoor O_3 levels were significant at $p < 0.01$, demonstrating that both outdoor and indoor levels affect personal O_3 exposure. Personal exposure is difficult to measure directly. This model may be useful to estimate personal exposure with indoor and outdoor O_3 levels.

Despite sample size limitations, personal exposure models were determined using microenvironmental concentrations. The objective of the modeling analysis was to investigate the feasibility of predicting personal O_3 exposures using a limited set of data. Limitations of this model

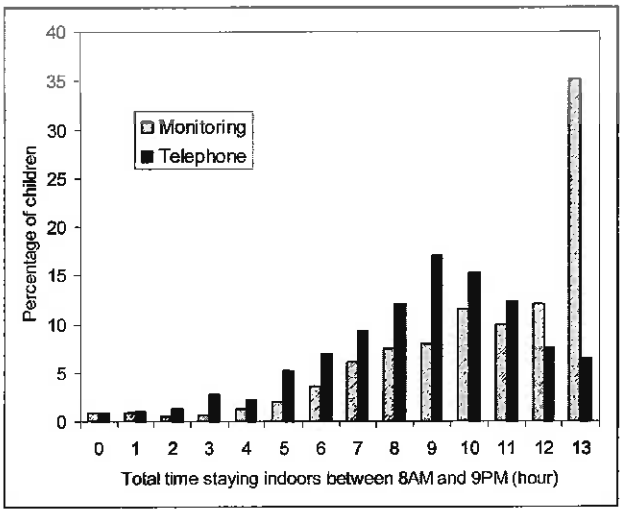

Figure 3. Comparison of total indoor times from monitoring study and telephone interview (8:00 a.m.–9:00 p.m.).

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may include the fact that only two microenvironments are incorporated, as well as the short averaging time (one week), leading to a loss of resolution in the correlation between times of high outdoor O_3 and the time of day children were outdoors. Both models showed statistically significant coefficients of indoor and outdoor O_3 levels, suggesting that outdoor levels as well as indoor levels affect personal O_3 exposure. Additional measurements in other geographic locations and microenvironments may help to improve the predictive power of the exposure models.

CONCLUSIONS

This study showed that a passive sampler could be used to measure outdoor/indoor/personal O_3 exposure levels of children. Participating children (4th and 5th graders) complied with the protocol over a 6-week period. The use of passive samplers can provide complex exposure profiles with high quality but at a low cost. The measurements using continuous monitors and passive samplers showed generally good agreement. However, the association can be influenced by characteristics of the stationary monitoring location. Personal exposures of children were between lower indoor and higher outdoor concentrations. Indoor O_3 concentrations are significantly associated with several housing characteristics (i.e., central air conditioning, window fan use, and open window), rather than being associated with outdoor O_3 concentrations. Having pets and spending time outdoors were significantly associated with personal O_3 exposures. Personal exposures of children who spent more time outdoors were higher than personal exposure of those who spent less time outdoors. Personal exposures were affected not only by the total time spent outdoors but also by the time of day the children were outdoors.

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ACKNOWLEDGMENTS

The authors sincerely thank the participants and their parents for a job well done. They thank the field staff: Camille Chandler, Kathy Chrestman, Mary Patton, Janet Orndorff, Rhonda Sagraves, and Cynthia Walton. In addition, Sarah Spengler and Maple Razsa assisted with telephone interviews. This work was sponsored by the Health Effects Institute, Contract No. 93–6. The authors acknowledge the assistance and guidance of the project officer Dr. Aaron Cohen, at the Health Effects Institute. The analysis described herein, in part, is sponsored by the Southern Oxidants Study (SOS)—a collaborative university, government, and private industry study to improve scientific understanding of the accumulation and effects of photochemical oxidants. Financial and in-kind support for SOS research and assessment activities is provided by the U.S. Environmental Protection Agency (EPA), National Oceanic and Atmospheric Administration, Department of Energy, Tennessee Valley Authority, National Aeronautics and Space Administration, EPRI, The Southern Company, Coordinating Research Council, and the states of Alabama, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, South Carolina, Tennessee, and Texas. Although the research described in this document has been funded in part by EPA under Assistance Agreement Nos. CR817732, CR818690, CR817766, CR818336, and CR824091, it has not been subjected to the Agency's peer and administrative review, and, therefore, may not necessarily reflect the views of the Agency, and no official endorsement should be inferred.

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Appendix C-2
Effect of Ventilation Strategies on Residential Ozone Levels

LBNL-5889E

ERNEST ORLANDO LAWRENCE
BERKELEY NATIONAL LABORATORY

Effect of Ventilation Strategies on Residential Ozone Levels

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August 2012

Submitted for publication to Building and Environment

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Acknowledgments

This work was supported by the California Energy Commission Public Interest Energy Research Program award number CEC-500-02-004 and the Assistant Secretary for Energy Efficiency and Renewable Energy, Building Technologies Program, of the U.S. Department of Energy under contract No. DE-AC02-05CH11231.

Abstract

Elevated outdoor ozone levels are associated with adverse health effects. Because people spend the vast majority of their time indoors, reduction in indoor levels of ozone of outdoor origin would lower population exposures and might also lead to a reduction in ozone-associated adverse health effects. In most buildings, indoor ozone levels are diminished with respect to outdoor levels to an extent that depends on surface reactions and on the degree to which ozone penetrates the building envelope. Ozone enters buildings from outdoors together with the airflows that are driven by natural and mechanical means, including deliberate ventilation used to reduce concentrations of indoor-generated pollutants. When assessing the effect of deliberate ventilation on occupant health one should consider not only the positive effects on removing pollutants of indoor origin but also the possibility that enhanced ventilation might increase indoor levels of pollutants originating outdoors. This study considers how changes in residential ventilation that are designed to comply with ASHRAE Standard 62.2 might influence indoor levels of ozone. Simulation results show that the building envelope can contribute significantly to filtration of ozone. Consequently, the use of exhaust ventilation systems is predicted to produce lower indoor ozone concentrations than would occur with balanced ventilation systems operating at the same air-exchange rate. We also investigated a strategy for reducing exposure to ozone that would deliberately reduce ventilation rates during times of high outdoor ozone concentration while still meeting daily average ventilation requirements.

Keywords: Ozone; mechanical ventilation; filtration; ASHRAE Standard 62.2; infiltration; simulation

1. Introduction

Ozone is an air pollutant that can have significant health effects. The most important source of ozone is outdoor air and most ozone inside homes comes from outdoor air entering the homes. Studies examining the relationship between ventilation rates and health find improved health with increased ventilation rates because most pollutant sources are indoors [1]. The effect of ventilation on ozone is reversed compared to internally generated pollutants because it is a pollutant from outside the homes. Because people spend the vast majority of their time indoors, reduction in indoor ozone levels could lead to improved public health in terms of ozone exposure. A summary by Weschler [2] showed that indoor ozone inhalation accounts for 25-60% of total ozone inhalation and inhalation of ozone reaction products are much greater indoors than outdoors. Weschler's summary also showed that the connections between ill health and the products of ozone reactions indicated that it would be better to prevent ozone entering home to reduce concentrations rather than relying on reactions within the home and that concentrations on the order of 10 ppb can lead to significant health effects. Some recent work has been done [3,4] on finding interior building materials that remove ozone from indoor air without producing harmful reaction products. This work also showed that the materials present in the building envelope are less likely to produce harmful ozone reaction products than interior surfaces because they are not coated with fatty acids associated with cooking and human skin lipids that react with ozone to produce volatile organic compounds.

The ozone entering homes from outdoors along with ventilation air is due to the natural forces of wind and indoor-outdoor temperature difference that induce infiltration and natural ventilation, operation of natural draft appliances and fans that are used to induce ventilation airflows mechanically. As air flows through a building envelope the ozone it contains can react with the surfaces over which it flows. Therefore, the quantity of outdoor ozone reaching the inside space may vary depending on the properties of the specific paths the air flow takes through the building envelope. Laboratory studies have indicated that ozone filtration by building envelopes can occur depending on ozone reactions with indoor materials [5]. Walker et al. [6] provide a bibliography of ozone reactions with indoor materials and the magnitude of this reduction is highly variable and ranges from zero to 90% and depends on the specific building materials that the air flows over and the geometry of the air flow paths through the envelope. This is linked to the effect that different mechanical ventilation systems have on the air flow paths for entry of air into a building. For example, compare a continuous supply ventilation system to a continuous exhaust ventilation system. For the supply system, the outdoor air does not pass through the envelope – instead air enters the building via a fan and its associated ducting. In this case, filtration may occur at dedicated outdoor air filters, or by deposition on the ducting used to deliver the air. For the exhaust system, air enters through leaks in the building envelope and leaves via a fan and its ductwork. In this case, any filtration of air entering the building occurs in the flow paths in the building envelope.

Another mechanical ventilation system characteristic is the ability to change the ventilation rate with time. In particular, the ventilation rate could be reduced during times of high outdoor ozone concentration and then increased at other times in order to meet the minimum requirements. The effect of lowering ventilation rates at peak outdoor ozone times was examined in this study by turning off ventilation systems at peak times. The intermittent ventilation requirements of ASHRAE Standard 62.2 were met by ventilating more during off –peak times.

This study quantifies, through modeling estimates, the impact on indoor ozone concentration of alternative residential ventilation strategies. Residential ventilation includes exhaust fans, supply fans and balanced systems. For exhaust fans, air entering the building passes through the building envelope and ozone is removed by reaction with building materials that line cracks and other points of entry. The supply and balanced systems do not have air entering the home via the building envelope flow path and so do not have the ability to remove ozone before it enters the homes. This study focuses on residential buildings and the particular envelope air flow paths and ventilation systems used in residences in the United States. The modeling has two parts. The first calculates the mass flow rates through individual building envelope air leakage sites and mechanical ventilation systems including the effects of wind and indoor-outdoor temperature differences. The second part calculates a mass balance for ozone that includes internal deposition and deposition in leaks that varies from leak to leak depending on the material properties and leak geometry. The outdoor ozone profiles were taken from California Air Resources Board (CARB) data [7]. The days with the highest ozone levels were selected as these will allow easier observation of differences between the ventilation systems. The particular days chosen were the peak ozone days for 2007: for Riverside this was August 12th and for Livermore this was July 22nd. Figure 1 illustrates the hour-by-hour outdoor ozone profile for these two days and locations – these concentrations are much higher than to 10 ppb discussed earlier that is considered to be high enough for health impacts to be an issue. The weather data used in the air flow calculations was taken from local weather stations for the same day as the outdoor ozone profile.

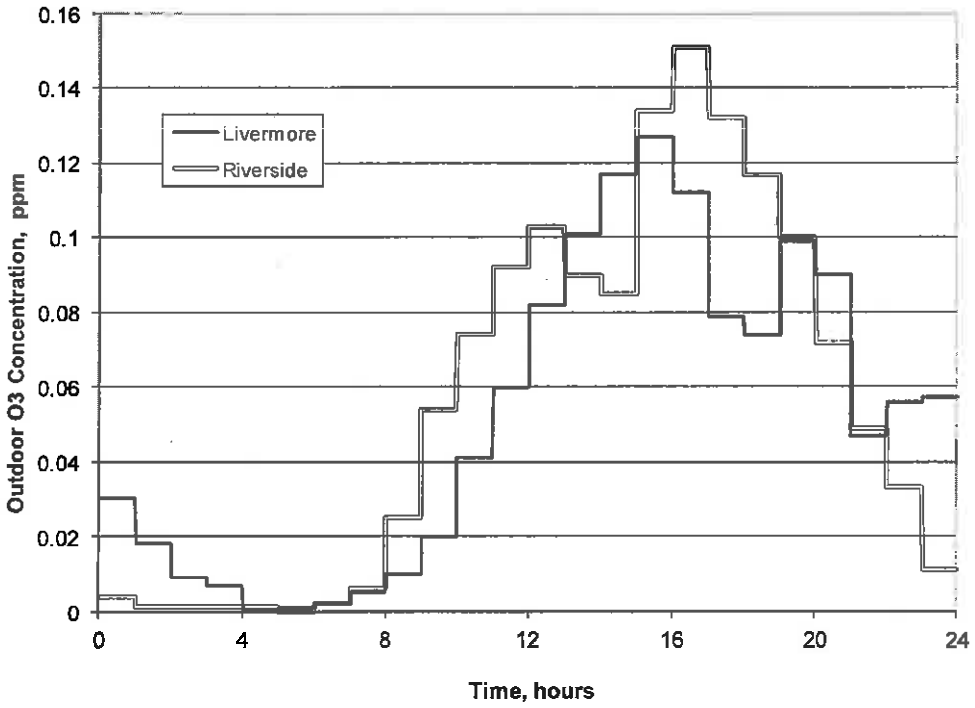

Figure 1. Outdoor ozone profiles for the peak ozone days in Livermore, CA and Riverside, CA.

There are four key ventilation/filtration parameters that determine the effect of the different ventilation options:

1. *Quantity of air flow.* Generally, less air flow means less ozone transported into the building. This must be balanced with the requirements of ASHRAE Standard 62.2 [8] that provides minimum whole house ventilation requirements for acceptable indoor air quality by diluting indoor pollutants with outdoor air. The mechanical ventilation systems in this study were sized to meet the ASHRAE 62.2 minimum requirements. In the case of intermittent ventilation the ASHRAE 62.2 intermittent ventilation calculation procedures were used to determine the air flows when the fans were operating. Similarly, the number of hours of credit during which mechanical ventilation is not needed due to economizer operation were also determined using the ASHRAE 62.2. Procedures. The air flow rates in this study were calculated using a ventilation, heat transfer and Heating, Ventilating and Air Conditioning (HVAC) equipment simulation tool called REGCAP that has been used in several previous studies of envelope air flows and their interactions with HVAC systems [9, 10, 11]. REGCAP was particularly suited to use in this study because it separates the air flow in and out of the house into specific leakage sites to which we can assign the different ozone penetration factors and thus evaluate the effect of using different air flow paths for outdoor ozone entry depending on the ventilation system used. The REGCAP tool combines a ventilation model, a heat transfer model and a simple moisture model. The ventilation model is a two-zone model: the attic and the house below it, and they interact through the ceiling flow. Both zones use the same type of flow equations and solution method. The total building and attic leakage is separated into components (such as 4 walls, ceiling, floor, flue, etc.). The flow for each component is determined by a power-law pressure - flow relationship with a flow coefficient, C, that determines the magnitude of the flow and an exponent for pressure difference, n, that determines how the flow through the leak varies with pressure difference. The ducts are modelled differently depending on if the air handler is on or off. When the air handler is off, the duct leaks are assumed to experience the same pressure difference as the ceiling. Air then flows between the house and the attic via these leaks. When the air handler is on, supply leaks enter the attic and return leak flows are form the attic to the return duct and there are register flows between the ducts and the house. The ventilation rate of the house and the attic is found by determining the internal pressures for the house and attic that balances the mass flows in and out. The heat transfer model determines the temperature of the house and attic air and other building components (e.g., pitched roof surfaces and ducts). A lumped heat capacity method is used to divide the attic, house and HAVC system into several nodes, and an energy balance including conduction, convection and radiation heat transfer is performed at each node to determine the temperatures. The simulation uses minute-by-minute time steps to account for cycling of HVAC equipment.

2. *Time of operation.* ASHRAE Standard 62.2 allows ventilation rates to change with time so long as various criteria are met. This allows for the possibility of reduced ventilation rates at times of high outdoor ozone concentrations, with increased ventilation at other times. Since ozone exhibits a strong diurnal profile, this strategy may offer a practical means to improve indoor air quality. Because the high ozone concentrations are coincident with high outdoor temperatures there is also the potential for reduced energy use.

3. *Air flow path.* This includes details of the air flow geometry for air flowing through the building envelope (including cracks, ducts, supply air filters, etc.) and relating envelope construction details to air flow path geometry and deposition. This changes the potential for deposition in the various air flow paths.

4. *Flow path materials.* Laboratory testing has shown that different materials react with ozone at different rates and generate different reaction products. The different reaction rates lead to changes in filtration potential. This study evaluated the effect of different mechanical ventilation systems on the air flow path and associated envelope materials.

A couple of factors will not be addressed in this paper because the mechanisms are poorly understood and the effects are not well quantified. The first is byproducts of ozone-initiated chemistry and the second factor is the time dependence of surface reactivity: aging and recovery. A more detailed discussion of these issues can be found in Walker et al. [6].

2. Calculations of indoor ozone concentration

The indoor ozone estimates were based on a single zone mass balance model that depends on air flows in and out of the building, the incidental filtration provided by individual air flow paths, deliberate filtration by the HVAC system, and deposition on interior surfaces. A single zone model was used because there are insufficient data available to describe the air flow and ozone characteristics of zone-to-zone interior flows making a multi-zone approach impractical. In addition, the comprehensive validation data required for a multi-zone model are not available. The following differential equation was used to describe the rate of change of indoor ozone concentration with time:

$$\frac{dC_{O_3,in}}{dt} = \left[\sum_i P_{O_3,i} Q_{in,i} C_{O_3,out}\right] - \left[Q_{out} + k + \eta Q_h\right] C_{O_3,in} \qquad (1)$$

Here, $C_{O3,in}$ is the indoor ozone concentration, $C_{O3,out}$ is the outdoor ozone concentration, $P_{O3,i}$ is the penetration factor of ozone for leak path *i* (including filtration for supply air), $Q_{in,i}$ is the volume-normalized air flow rate into the house (in air-changes per hour, h^{-1}) through flow path *i*, Q_{out} is the volume-normalized air flow rate out of the house (h^{-1}), Q_h is the volume-normalized air flow rate through the HVAC system (h^{-1}), *k* is the indoor deposition loss-rate coefficient (h^{-1}), and η is the single-pass ozone removal efficiency in the HVAC system. Based on data in Hyttinen et al. [20], we set the value of η to be 5%, whenever the HVAC system is on. The air flows were assumed to be quasi-steady over the short time period of one minute for the simulation steps. Equation (2) was used as a discrete approximation of equation (1) for evaluating the time-varying level of indoor ozone, with Δt = 1 min = 0.0167 h:

$$C_{O_3,in}(t) = C_{O_3,in}(t-\Delta t) + \Delta t \times \left(\left[\sum_i P_{O_3,i} Q_{in,i}(t) C_{O_3,out}(t)\right] - \left[Q_{out}(t) + k + \eta Q_h\right] C_{O_3,in}(t-\Delta t) \right) \qquad (2)$$

In many field studies results are expressed as indoor:outdoor concentration ratios. These field data will be used as a validation of the modeling/simulation approach, therefore the concentrations calculated using the above technique were also expressed as a fraction of the coincident outdoor concentration. In rough terms, not accounting for envelope deposition or HVAC system removal, the indoor:outdoor ratio of the time averaged concentrations is given by Equation 3 , where *Q* is the air-exchange rate for the house and *k* is the deposition factor.

$$\frac{C_{O_3,in}(t)}{C_{O_3,out}(t)} = \frac{Q}{Q+k} \qquad (3)$$

3. Ozone removal by the building envelope

The extent of ozone penetration through building envelopes depends on air flow velocity, crack geometry, and reaction probabilities on the surface. The relationships describing ozone deposition calculations for the building envelope were based on previously published work [5, 21]. In this study we focused on applying these models to common air leakage paths for homes in a way that enables us to estimate indoor concentrations. Different performance parameters related to ozone removal were assigned to the different leakage paths in the building envelope depending on their geometry and construction materials.

The removal of ozone by the building envelope was separated into two main categories corresponding to different air flow paths: flow through cracks and flow through insulation. The floor level leakage was assumed to flow through cracks, and the wall and ceiling flows are through insulation. We further separated the air flow entering the building through cracks into flow through cracks in the walls and ceiling, and flow through floor levels leaks where one surface is the concrete foundation slab.

The mean velocity, $\overline{V}$, of air flow in a crack is given by Equation 4:

$$\overline{V} = \frac{Q}{A} \tag{4}$$

where Q is the volumetric air flow rate and A is the cross-sectional area of the flow path.

The relationships used to determine ozone penetration in cracks were taken from the analysis given in [5]. The penetration factor (P_{O3}) for ozone in cracks was given by:

$$P_{O_3} = \exp\left(-\frac{2V_o L}{\overline{V} d}\right) \tag{5}$$

where V_o is the deposition velocity and d is the crack height (m).

To calculate ozone deposition in cracks we used the relationships developed in [5]. We used the following values for ozone-surface reaction probability — For glass fibers: $\gamma = 6\times10^{-6}$ for freshly exposed glass fibers [5] - with a surface uptake deposition velocity of 5.4×10^{-4} m/s; For concrete: $\gamma = 4.4 \times 10^{-5}$ to 10^{-4} for outdoor surfaces and 7.9×10^{-5} for a slab [28]. Given this range a central value of 5×10^{-5} was selected for concrete-faced cracks. For plywood (using the same value for wood surfaces): $\gamma = 4.7 \times 10^{-6}$ to 5.8×10^{-7} [18]. A central value of 1×10^{-6} was used in the calculations.

More recent work has included more thorough measurements for wood and concrete and gives values of surface uptake deposition velocities directly:

- 0.00080 m/s for coarse concrete and 0.00017 m/s for fine concrete at 50% RH [29].
- 0.00022 m/s for untreated unfinished wood at 50% RH [29].
- Deposition velocity for gypsum wallboard is 0.0012 m/s [30].

Grøntoft [31] found a reduction in deposition velocity for successive tests of an individual sample. This implies that the deposition rate will vary depending on the deposition history of the surface. The decrease was about 20% for concrete floor tile. Given the paucity of data on the reduction in deposition velocity, particularly for the materials of interest in this study, this effect was ignored. In addition, there are sure to be variations in deposition velocity with temperature, and, again because of the lack of

necessary data, these effects were ignored. The values used in the current study are those at steady state.

For all but the largest concrete faced cracks, the air flows and crack geometries of interest in this study result in deposition velocities that are limited by the surface-uptake deposition velocity that is typically an order of magnitude lower than the mass transport velocity. For example, a wood faced crack at 0.1 m/s has a surface uptake velocity of 0.00023 m/s and a mass transport velocity of 0.007 m/s.

For a flow path through a wood-faced envelope crack, the ozone penetration for three different crack heights is shown in Figure 2. The results in Figure 2 show that for the larger leak characteristic of 10 mm the penetration factor is greater than 95% under typical conditions (air velocity > 0.1 m/s). For concrete, the higher reaction probability results in less ozone penetration, as shown in Figure 3.

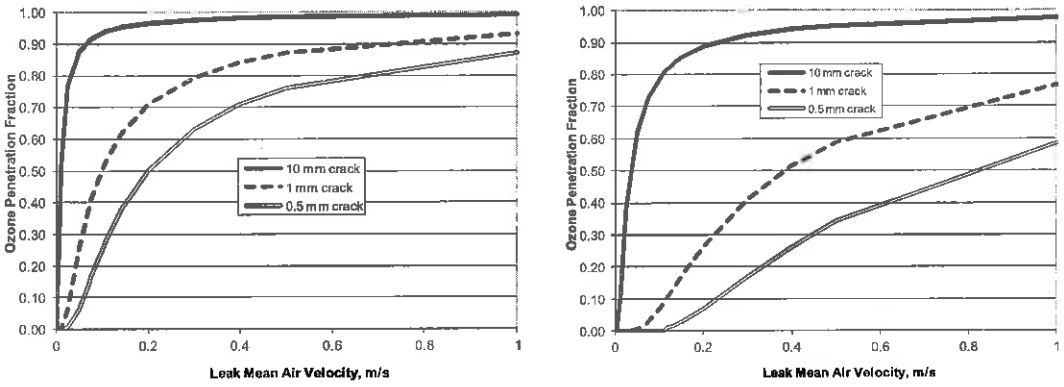

Figure 2. Ozone penetration in 0.15 m long wood faced (left) and concrete faced (right) cracks.

The results for 0.15 m long paths are appropriate for air flow paths that do not flow through the cavity insulation. For flow through the wall cavity and the ceiling, the air flow path also includes about 1 cm of gypsum wallboard on the interior surface and about 1 cm of exterior sheathing - commonly plywood. Figure 3 shows that the short crack has essentially total penetration at velocities greater than 0.1 m/s, indicating that the deposition in plywood and gypsum board sheathing can be neglected and the deposition inside wall cavities and ceiling insulation dominates for these flow paths.

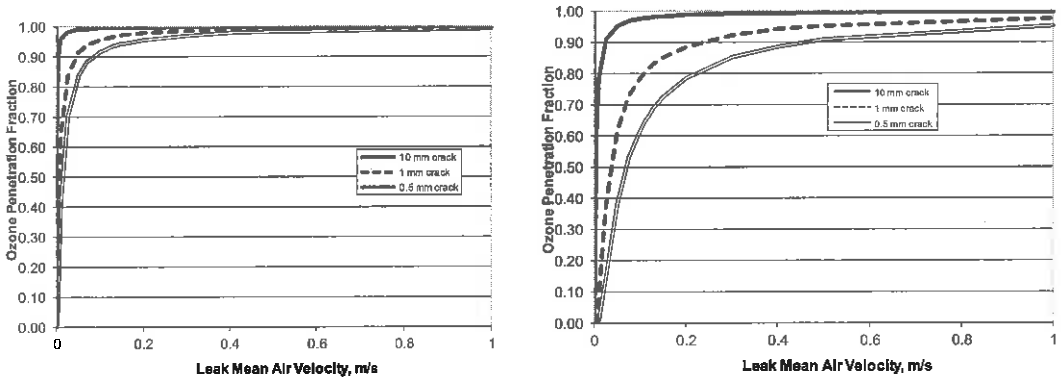

Figure 3. Ozone penetration for a short 0.01 m long wood faced (left) and gypsum faced (right) crack.

The crack at floor level will have one face that is wood and the other that is concrete. Thus, the two faces of the crack have different uptake coefficients. A previous analysis [5] reported the basic work on this topic, but limited their analysis to cases in which the crack walls had common reaction rate coefficients. The following analysis determines the relationships for a crack with two different faces. The flow through the crack is represented in two dimensions, with the top and bottom surfaces lining the flow channel. A steady-state material balance equation is written on ozone over an along-stream segment of thickness Δx. If the rate of deposition uptake is the same on both surfaces, then the material balance would be written as follows:

$$\overline{V}dWC\left(x-\frac{\Delta x}{2}\right)=\overline{V}dWC\left(x+\frac{\Delta x}{2}\right)+2V_o\Delta xWC(x) \tag{6}$$

In equation (6), the first term on the left represents advective transport into the control volume. The terms on the right represent, respectively, advective transport out of the control volume and depositional loss on the top and bottom surfaces. The "deposition velocity" model was used for surface loss, where the flux to a surface is equal to $V_o C$, with V_o being the deposition velocity.

To combine the effects of mass transport and surface reaction, we follow the previous approach of Liu and Nazaroff [5] and create a linear model of the deposition process in which the mass transport and surface reaction process act in series. The overall flux to either surface is $2V_oC$, and series process treatment produces this relationship:

$$V_o=\frac{V_sV_t}{V_s+V_t}=\left(\frac{1}{V_s}+\frac{1}{V_t}\right)^{-1} \tag{7}$$

Where V_s is the surface reaction velocity and V_t is the mass transport velocity. Allowing for different surface reaction rates for two different crack surfaces (e.g., wood and concrete) that correspond to V_{s1} and V_{s2}, we can write the combined expression for the dependence of the overall deposition velocity on the surface uptake limited deposition velocities and the transport-limited deposition velocity.

$$V_o=\frac{V_t\left(V_{s1}V_t+2V_{s1}V_{s2}+V_{s2}V_t\right)}{2\left(V_{s1}+V_t\right)\left(V_{s2}+V_t\right)} \tag{8}$$

The surface uptake-limited deposition velocities are estimated knowing the total deposition velocities (0.00022 m/s for wood and 0.00080 m/s for concrete) given by Grøntoft and Raychaudhuri [29] at particular transport deposition velocities. For wood, V_s = 0.00022 m/s (the same as V_0) and for concrete, V_s = 0.0009 m/s. These values of V_s were used together with V_t calculated for each time step in the simulations in order to obtain V_o for each time step, and thereby determine ozone penetration using Equation 5. Figure 4 shows the resulting ozone penetration for a two faced crack, with one face of wood and the other of concrete.

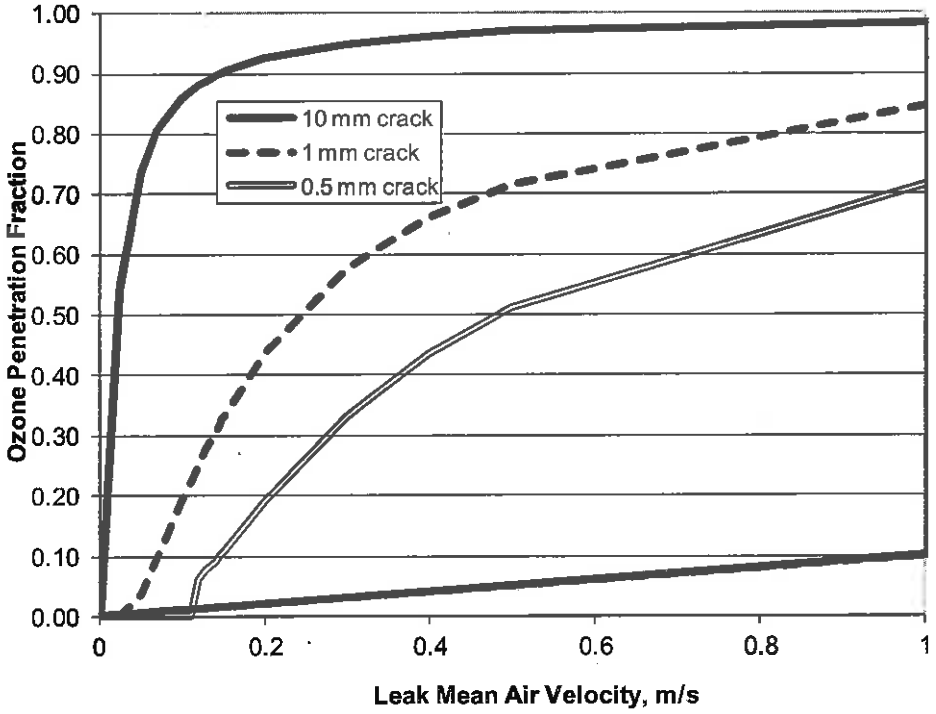

Figure 4. Ozone penetration for a 2 faced crack - one face is wood, the other concrete.

The wall and ceiling leaks both have air flows through glass fiber insulation in addition to having leaks through or around the solid components. For the walls, the insulation is in a cavity where the inner cavity face is gypsum wallboard and the exterior face is plywood sheathing. The ozone filtration in insulation is different from that through cracks and depends on the glass fiber diameter, fiber packing density (solidity), mean air velocity and flow path [5]. For glass fiber insulation, the fiber diameter is well known and is assumed to be the same for all the insulated cavity air flow paths. The ozone removal is then determined by the geometry of air flow through the insulation. The fraction of total air flow through insulation for wall and ceiling leaks is given by the parameter "*f*".

For wall cavities, there are two extremes of cavity flow. One is flow that is straight through the wall (horizontal flow) and has a path length of the wall thickness. The second is vertical flow through the cavity, where the path length is the full cavity height. In our calculations, we assumed typical wall dimensions with a wall thickness of 0.15 m and a cavity height of 2.5 m. The vertical flow in wall cavities has a much greater path length and will have almost total ozone removal; however, this flow path has a high air flow resistance and only a small fraction of the total envelope air flow will take this path (in our example the path length is 17 times greater than for the horizontal flow and therefore a factor of 17 less air flow), that reduces the potential for ozone removal. In other words, this vertical flow path will have a lot of ozone removal from only a small fraction of the air entering the building, resulting in little change in overall envelope ozone removal efficiency. Therefore, the calculations in this study only include the horizontal flow. This yields penetration factors similar to the range of 0.62 to 1.0 (with a mean of 0.79) found in a field study estimation of penetration factors [19].

The ozone penetration through a wall section with fiberglass insulation is given by:

$$P_{O_3} = f \times \exp\left(-\frac{4V_o \alpha L}{\bar{V} d_f}\right) \tag{9}$$

where: α is the solidity of the insulation (about 0.003 for glass fiber) and d_f is the fiber diameter (about 10×10^{-6} m or 10 μm). L is the flow path length that changes for wall and ceiling air flow paths. Figure 5 shows the ozone penetration fraction for the three different glass fiber filled air flow paths with f = 1 (i.e., all flow goes through the insulation).

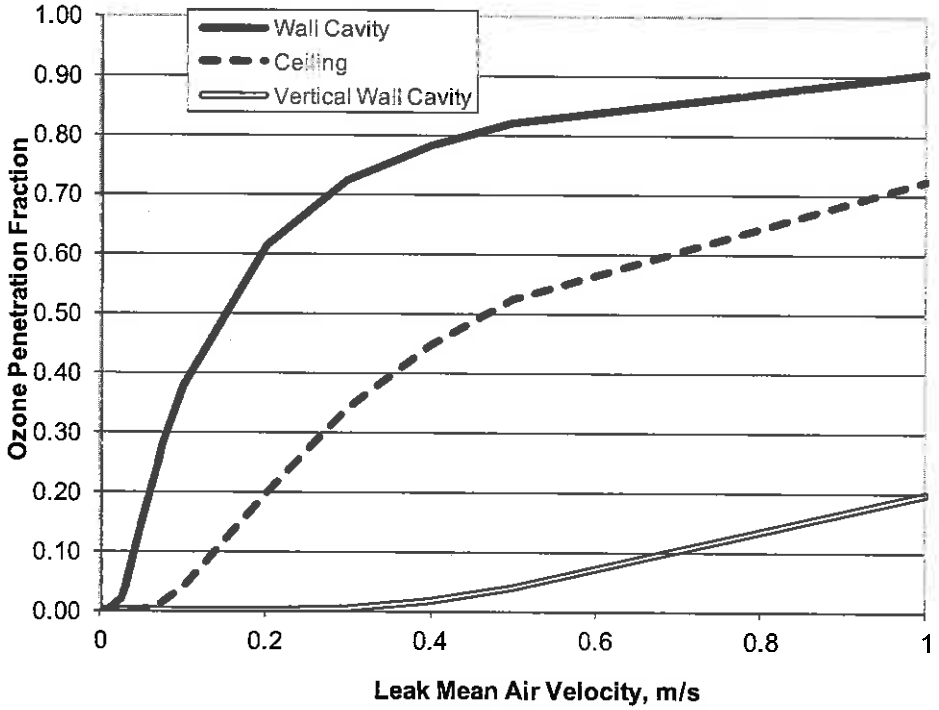

Figure 5. Ozone penetration factors through glass fiber insulation for a 0.15 m wall cavity, a 0.5m thick ceiling insulation blanket, and a 2.5 m tall vertical wall cavity flow path.

To determine the ozone penetration, the air velocity through the insulation is required. We determined the mean air velocity for air flowing through the insulation using the geometry of the air flow, where the mean air velocity is given by Equation 2 using the volumetric air flow rate (from the air flow model) and the cross-sectional area of the flow path. The mean velocity was determined for three different air flow paths:

1. The 0.15 m straight through wall gives a mean velocity of 0.0009 m/s,
2. the 0.5 m of ceiling insulation has a mean velocity of 0.00027 m/s, and
3. the 2.5 m vertical flow path through a wall cavity has a mean velocity of 0.0005 m/s.

At these low velocities, there is almost zero penetration of ozone.

Before assuming that we could assume zero penetration of ozone, we recalculated the velocities at a higher leak pressure of 5 Pa, in which case the air flows increase to 0.0046 m/s, 0.00138 m/s and

0.00028 m/s for the same three air flow paths. These air velocities are still not high enough for there to be any ozone penetration. These calculations imply that penetration can be assumed to be zero all the time for air flow through the glass fiber insulation in the ceiling, so ceiling air flows will not transport any ozone into the building. Therefore the critical parameter determining deposition rates for ceiling air flows is f. Reasonable values for indoor concentrations compared to measured field data can be obtained by assuming half of the inflowing ceiling air flows bypass the insulation, i.e., $f = 0.5$ [6]. With no bypass the resulting indoor concentrations are too low compared to the measured field data.

The floor level leaks have two faces: the wood of the framing and the concrete of the slab/foundation. Each face has its own ozone reaction properties, and the expression for deposition velocity (V_o) for a two-faced crack given by Equation 8 was used, with fixed values of $V_s = 0.00022$ m/s for wood and 0.0009 m/s for concrete. V_o was used in Equation 9, together with crack geometry (10 mm height and 0.15 m length) and air velocity to estimate the ozone penetration. Each of the four floor level leaks has its own air flow, air velocity and ozone penetration factor because the air flow through each of the four floor level leaks is different – and may not even be in the same direction. The relationship between air flow velocity, crack dimensions and the leak pressure exponent is based on the methods developed by Sherman [27]. Assuming a typical leak pressure of 2 Pa and a crack length of 0.15 m the crack area is 0.000082 m^2 corresponding to a crack diameter of 10 mm and the mean air speed is 0.6 m/s.

4. Field measurements of ozone in homes

The results of field measurements from other studies were used for two purposes in this study. The first was to determine typical ratios of indoor to outdoor ozone concentrations as a check on the simulation predictions. The second was to determine input values for parameters used in the simulations. The ratio of indoor to outdoor ozone concentrations has been measured in several studies: An extensive summary of indoor-outdoor ozone concentration ratios measured in about 60 studies covering a wide range of buildings was reported in Weschler [2]; indoor to outdoor concentrations of 0.37±0.25 for 136 were found in southern California homes [12]; and indoor to outdoor ratios of 0.20±0.18 in 145 Mexico City homes [13]. A complication in interpreting the indoor to outdoor ozone concentration ratio in these studies was that many of the homes had open windows during sampling. Homes with open windows will have no ozone removal as air flows through the window opening and thus result in higher indoor concentrations compared to homes with closed windows. Data from the National Morbidity, Mortality, and Air Pollution Study have been correlated with US Census and American Housing Survey data, and showed reduced mortality due to ozone exposure and the presence of central air conditioning in homes (which are likely to be homes with less window opening) [14], with the caveat that the presence of central air conditioning is highly correlated with higher temperatures, i.e., when window opening is less likely to be used for comfort. Focusing on homes with closed windows several studies report that the resulting indoor-outdoor concentration ratio is about 0.1 [15, 16, 17]. These measured values will be used as a check on the simulations. Given the large ranges found in these studies, a target range of indoor to outdoor ozone concentration ratios is about 0.05 to 0.4. So long as the ratios are within this reasonable range, this study will then concentrate on the relative changes in indoor concentration depending on the ventilation technique employed.

In addition to indoor:outdoor ratios, some studies have determined the deposition factor, k, in Equation 3 and are summarized by Weschler [2]. The most relevant for houses were for 43 homes with a mean deposition factor over all the houses of 2.8 per hour – with significant variation from house to house leading to a standard deviation of 1.3 per hour [15]. So, the 0.05 to 0.4 range of indoor to outdoor

ozone concentration ratios corresponds to air-exchange rates in the range 0.15 to 2 per hour if envelope deposition is ignored. Other field studies [2, 18] showed values of interior deposition that changed significantly with HVAC system operation: the interior deposition coefficient changing from 2.9 (similar to the 2.8 from [15]) for no HVAC operation to 5.4 with HVAC operation. To account for this effect in our simulations we will include the effects of increased ventilation rates due to HVAC system air leakage and ozone deposition on HAVC system filters.

Overall, these field measurements suggest that interior deposition is a significant contributor to reductions of indoor ozone concentration. This has the effect of lowering the mean values of indoor ozone concentrations. So, although this study focuses on the differences in indoor concentration due to the operation of different mechanical ventilation systems, primarily due to envelope deposition, it is important to realize that interior deposition has a significant effect on the indoor ozone concentration. A recent study [19] measured ozone penetration in eight houses by measuring the decay of artificially elevated ozone levels simultaneously with fan induced ventilation. The results showed a mean ozone penetration rate of 0.79 (ranging from 0.62 to 1.0) indicating that ozone removal as air travels through the envelope is significant. Combined with a measured indoor deposition rate of 9 per hour (much higher than other estimates primarily due to operating mixing fans during the experiment) the estimate of indoor:outdoor concentration was 0.05, which is in line with other estimates. The authors concluded that this ozone removal as air flows through the envelope is a "preferred first line of defense against indoor exposures to outdoor ozone and its byproducts".

5. HVAC system filtration

To account for the filtration of ozone provided by HVAC system filters, it was assumed that the filters are dirty (as these are the most likely to be in place in residences.) Dirty filters remove 3 to 20% of the ozone [20]. A typical value of 5% of ozone removal was used for central forced air system filter and for the filters on the supply side of HRV and continuous supply systems.

6. House characteristics

Houses were simulated for two climate zones that had high outdoor ozone levels: Livermore, CA, and Riverside, CA. The house characteristics were based on a California State Energy Code [22] prototype home that is a slab-on-grade, two-story, with a floor area of 164 m^2. The wall surface area was estimated from measured data from several thousand new homes [32]: the wall area was typically 1.54 times the floor area for a two-story home. Window area was 20% of floor area with windows equally distributed on the four exterior walls.

The assumed total envelope leakage was chosen as a typical value for new home construction and is the default in California's residential building energy code [22]: Specific Leakage Area of 4, corresponding to a flow coefficient (C) of 0.067 $m^3 s^{-1} Pa^{-n}$. The envelope pressure exponent was assumed to be n = 2/3 and was assumed to be the same for all leaks and independent of flow rate based on a large national database [23]. The pressure exponent has been shown to be constant over the pressure and flow range of interest [24]. The wind pressures included local shielding by adjacent buildings based on the wind shadow shelter method of Walker et al. [25]. The houses were assumed to be in a typical urban environment with houses in a row on a street. The envelope leakage distribution was assumed to be 15% at floor level, 50% in the walls and 35% in the ceiling based on default values in the ASHRAE Handbook of Fundamentals [26]. The wall and floor level leakage was split into four equal parts – one

for each face of the building. The heating and cooling systems were modeled to include air leakage between the ducts and attic at 5% of total system flow (2.5% supply leaks and 2.5% return leaks).

7. Ventilation systems

In addition to the ventilation systems outlined below, the house had a bathroom exhaust that operated from 7:30 to 8:00 a.m. (two 25 L/s fans) and a kitchen exhaust that operated from 5:00 p.m. to 6:00 p.m. (an 80 L/s fan). The simulations evaluated the following ventilation systems that are all ASHRAE Standard 62.2 compliant (except for open windows) and commonly used in residences:

1. *Continuous exhaust.* This continuously depressurized the house, and the incoming air enters through the building envelope. Depending on envelope leakage and weather conditions, the flow path for air entering the home could change (e.g., between floor, ceiling or wall leaks), and with it the ozone removal.
2. *Intermittent exhaust.* In addition to the characteristics of the continuous exhaust, this system had the capability to be turned off at times of high outdoor ozone levels. This can be done by simple timers, in response to an alert, or, conceivably, through internet-based information linked to real-time local ozone reporting. In the simulations, a timer was used to turn off the exhaust system for four hours at a time. In summer, the off period was from 3:00 p.m. to 7:00 p.m. (coincident with the electrical system peak load and also avoiding some periods of high ambient ozone). In winter, the off period was from 1:00 a.m. to 5:00 a.m. The intermittent exhaust was sized to meet ASHRAE 62.2 and, therefore, had 20% more air flow than the continuous exhaust.
3. *Heat Recovery Ventilator (HRV).* The HRV had balanced air flow which meant that the envelope flows were generally unchanged during operation.
4. *Central Fan Integrated Supply (CFIS) with continuous exhaust.* This system utilized both supply and exhaust, with the supply operating for 20 minutes out of each hour in conjunction with the central furnace or air conditioning blower.
5. *Continuous supply.* This system continually pressurized the house and incoming air enters mostly through the fan and exits through the building envelope. Depending on envelope leakage and weather conditions, the exact flow path could change, and with it the ozone removal.
6. *Economizer.* An economizer was modeled as a supply fan with a flow set equal to the HVAC central system blower at cooling speed. The economizer operates when the indoor temperature is above 21 °C and the outdoor temperature is 3.3 °C or more below the indoor temperature setpoint. Although the economizer has very high airflows, the long off times mean that it is not possible to be ASHRAE 62.2 compliant with the economizer alone. Relationships have been developed between high and low ventilation rates and their respective operating times [33] that indicate that for the long (16 hours for Climate Zone (CZ) 10) off times there is no economizer air flow rate that would allow this system to meet ASHRAE 62.2 over a 24 hour period. However, it is possible to take credit for this over-ventilation. Credit can be taken for over-ventilation/economizer operation (i.e., time during which no mechanical ventilation is required) [9]. In this case of roughly 6 hours of economizer operation there is 10 hours of credit available. To minimize ozone entry, it makes sense to take this credit between the hours of 10:00 a.m. and 8:00 p.m. This means that that additional mechanical ventilation occurs when the economizer is off before 10:00 a.m. and after 8:00 p.m. During these times, the outdoor ozone concentrations are low and will not significantly contribute to the indoor ozone concentration.
7. *Open Windows.* In previous studies and summaries [12, 2], it has been shown that open windows significantly increase indoor ozone levels when outdoor ozone levels are high. This

outcome is understood to be due to a combination of significantly (at least by factors of two) increased ventilation rates and zero filtration of entering air.

In addition to the whole building ASHRAE 62.2 compliant ventilation systems, the house had a bathroom exhaust that operated from 7:30 to 8:00 a.m. (two 24 L/s fans) and a kitchen exhaust that operated from 5:00 p.m. to 6:00 p.m. (a 75 L/s fan).

8. Ventilation results

An example of the incoming ventilation air flow rates are shown in Figure 6 for the continuous fan, where a positive value for flow is flow into the house and a negative value is flow out of the house. The figure shows the flow through the ceiling and inflows only through wall and floor leaks (each of the four wall and floor leaks were summed to get the building total). These air flows are in kg/s. To convert to Air Changes per Hour (ACH), one must multiply by 6.8: e.g., the highest air flows in Figure 6 (when the kitchen exhaust is on) correspond to about 0.95 ACH. The effect of the morning/evening bathroom and afternoon kitchen exhaust stand out in these results. From 2:00 p.m. to 6:00 p.m. the duct leakage increases the ventilation rate by about 0.02 kg/s (0.14 ACH). The cooler indoor temperatures during the daytime result in inflows through the ceiling and outflows through the floor - resulting in no inflow through the floor. The continuous supply pressurizes the house resulting in less inflow for other envelope flow paths. In this case, the ceiling rarely has any inflow, as shown in Figure 7. The variability of flow magnitude and direction was used as input to the indoor ozone calculations. It was important to separate the leaks in this way and know which leaks were contributing to air flow into the building (and the magnitude of the flows in these leaks) so that the different ozone removal rates for each leak could be applied.

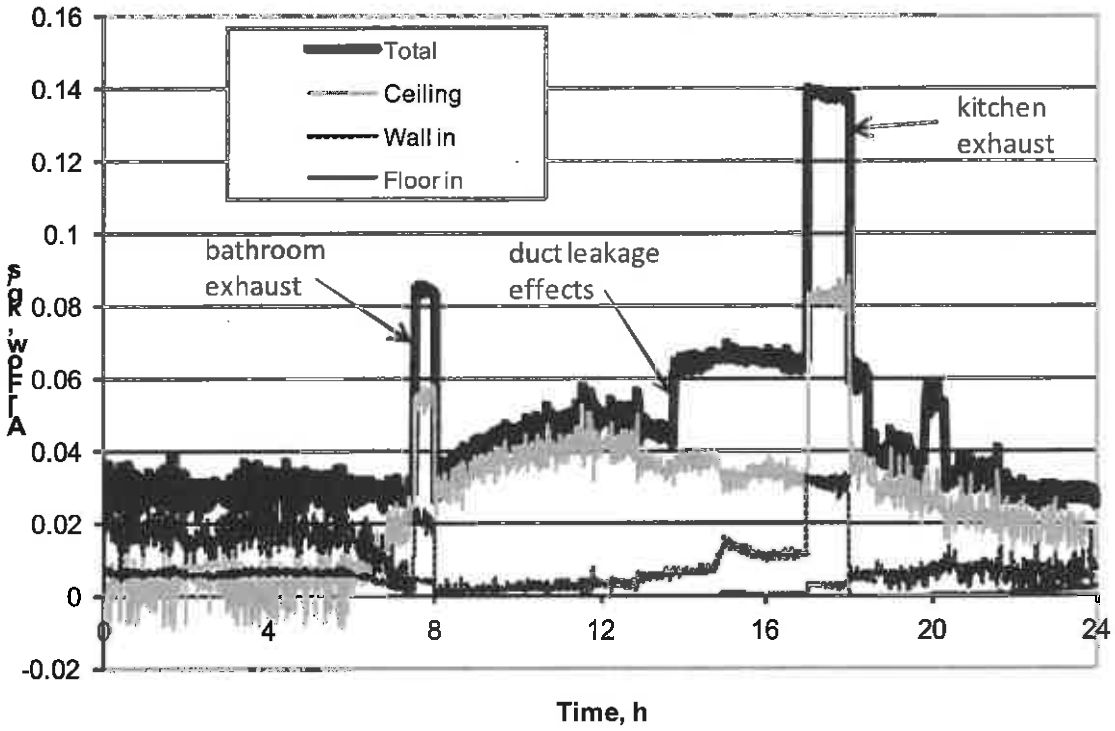

Figure 6. Envelope air flows for continuous exhaust in Riverside.

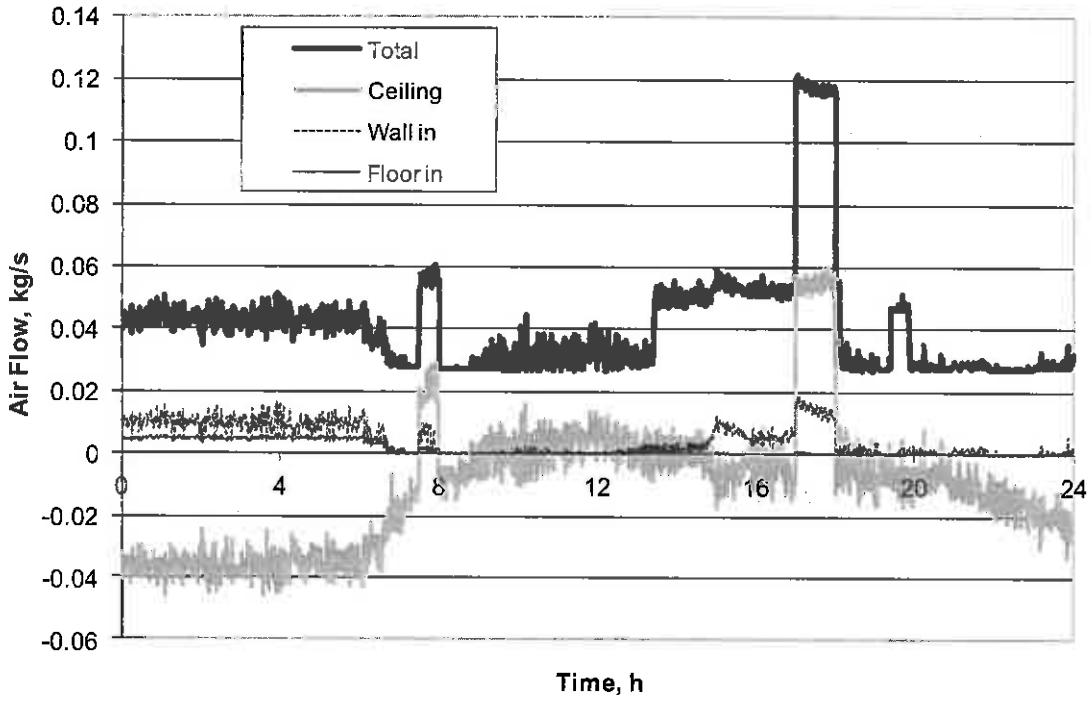

Figure 7. Envelope air flows for Continuous Supply in Riverside.

9. Results and Discussion

Some additional simulations were performed in order to provide context for the envelope filtration results by assuming no envelope filtration and looking at the effect of open windows for comparison to field data to further verify model predictions. With no envelope filtration the internal deposition combined with a small amount of filtration from the HVAC system filters brings down the indoor ozone concentrations by about 80%. Measured field data for houses with normally open widows had indoor:outdoor concentration ratios averaging 0.68±0.18 [15]. To see if the simulations can replicate this result, a simulation was run for Riverside using open windows - one on the first floor and one on the second floor. Each window had a 1m high by 0.5 m wide opening. The results were that the ratio of average indoor:outdoor concentration is 0.47 (the average of the ratios is 0.40 and ratio of peaks is 0.58). The big increase in ventilation results, as expected, in large increases in interior ozone. Without knowing more about the weather or window opening details in the measured field data, the results are reasonable. Figure 8 shows these effects relative to each other for the case of continuous exhaust.

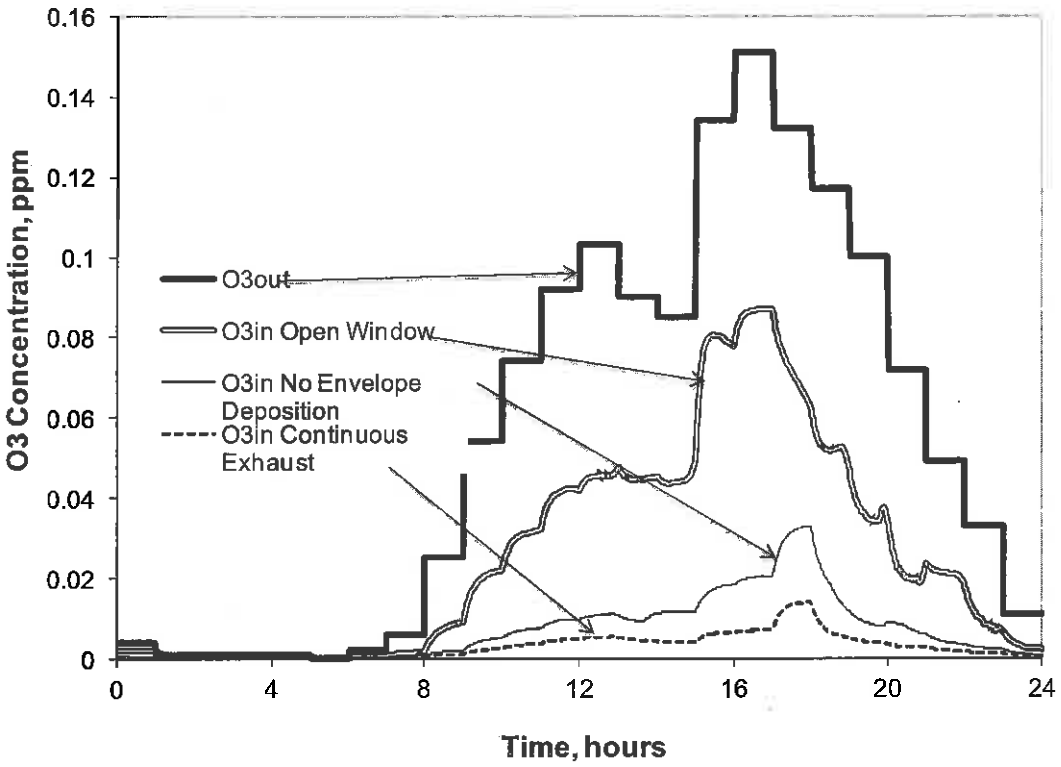

Figure 8. Effect of open window, no envelope deposition and inclusion of envelope filtration (for the case of continuous exhaust) for Livermore

Figure 9 compares the outdoor concentrations to indoors for the continuous exhaust in Riverside. As a check on the validity of the results, the concentrations shown in Figure 9 for Riverside with continuous exhaust can be compared with field measurements of Lee et al. [15]. Figure 9 shows that the indoor concentrations generally follow the outdoor ones – but at a much lower level. The case shown in Figure 9 is for continuous exhaust that maximizes ozone reductions due to interactions with the building envelope and represents the best case for envelope ozone removal resulting in an average indoor:outdoor concentration ratio of 0.053. This best case situation is at the low end of field data.

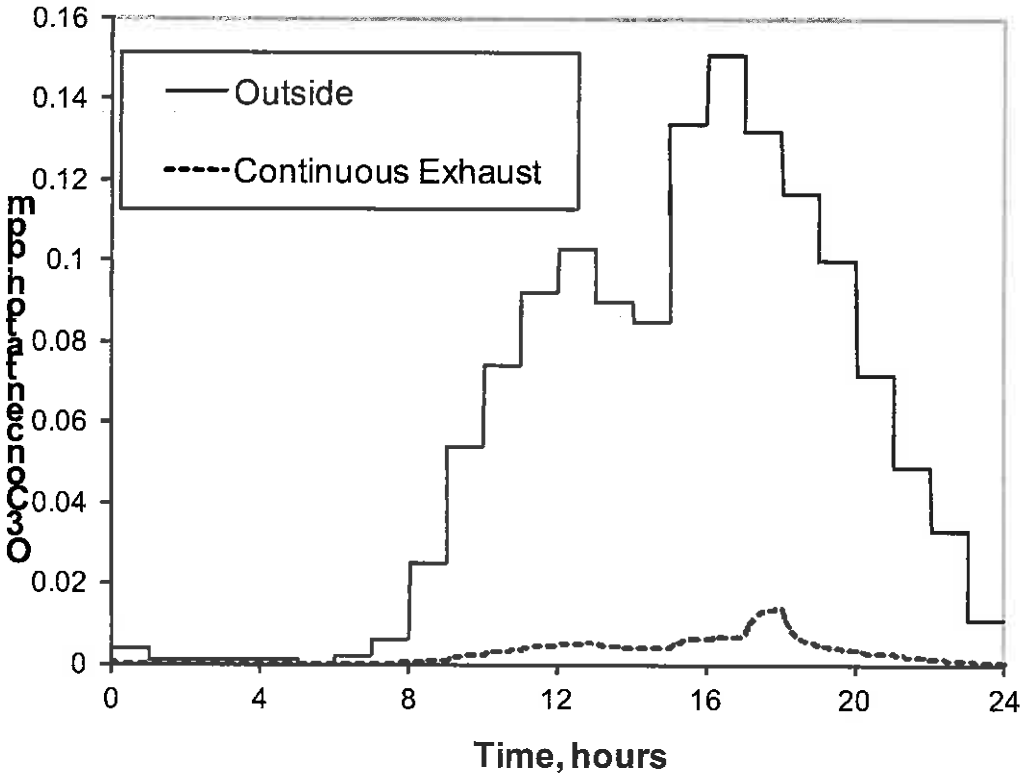

Figure 9. Indoor and outdoor ozone concentration in Riverside for continuous exhaust.

Figure 10 compares the indoor ozone concentrations for all the ventilation system alternatives in Riverside (the results for Livermore are essentially the same – just with a slightly different profile due to the different outdoor ozone profile). The spike upward in ozone concentration for all the systems at hour 17 is due to the operation of the kitchen exhaust fan, as seen in Figures 6 and 7. The economizer has the lowest concentrations due to having no mechanical ventilation for ten hours of the day. The intermittent exhaust diverges from the continuous exhaust and has lower concentrations when it is off. The cycling of the CFIS and HRV systems are clearly observed. The higher air flow rates when operating lead to the higher peaks for these systems.

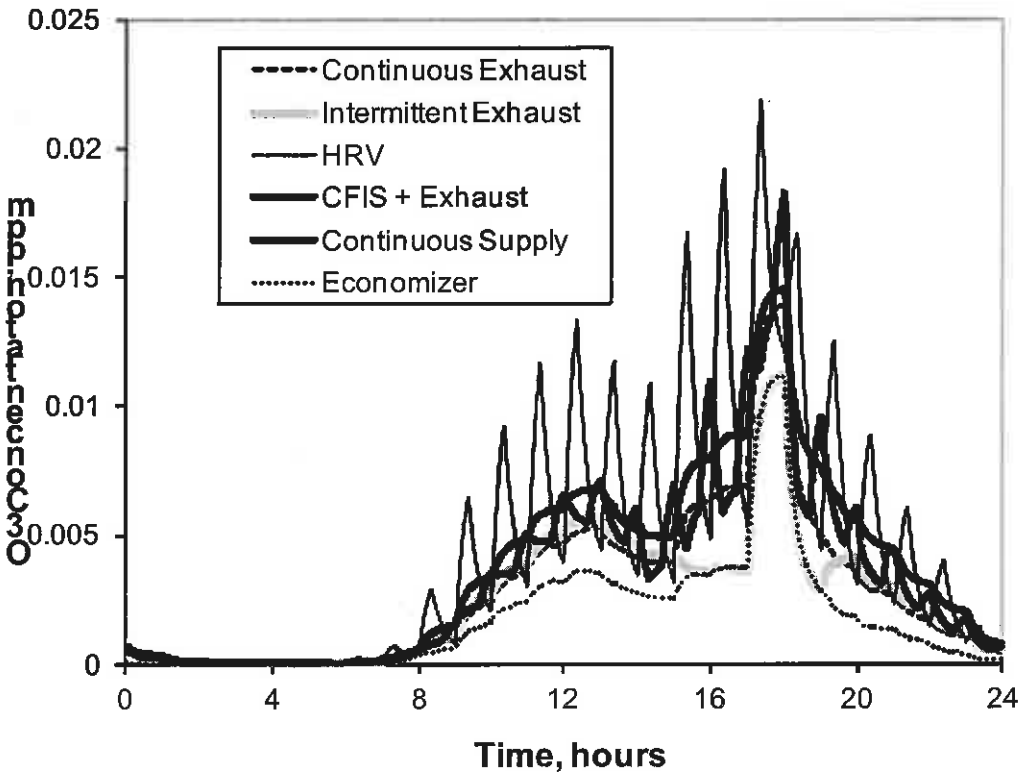

Figure 10. Comparison of indoor ozone concentration for different ventilation strategies in Riverside.

For each ventilation system, the average air change rate was calculated, together with the average indoor ozone concentration, the average indoor:outdoor concentration ratio (all over a 24 hour period) and the indoor peak. Table 1 summarizes all these results and shows that for all but the open windows case, the average indoor concentrations are below the 10 ppb that is considered a level of concern, and the better performing (from an ozone level perspective) ventilation systems have peaks that are close to 10 ppb. For comparison purposes, the outdoor peak and average concentrations are: 56 and 151 ppb for Riverside and 52 and 127 ppb for Livermore. Table 1 shows that the HRV produced the highest indoor ozone concentrations. This is due to a combination of having a higher air exchange rate then the unbalanced ventilation cases because balanced ventilation combine linearly adds with the natural infiltration through envelope leaks, but the supply and exhaust system air flows combine sub-additively. This higher air exchange brings more ozone into the home. The HRV and supply systems had higher peaks concentrations than the exhaust system because, while they had the same average mechanical air flow, they cycled on and off and had higher airflows when their fans were turned on. The intermittent exhaust deliberately ventilated less when the outdoor concentrations of ozone were highest, and this resulted in about a 10% reduction in indoor ozone concentrations. To be more useful, this strategy would need to be off for a longer period of time, or be used in a building with a tighter envelope where the natural infiltration with the mechanical ventilation system turned off would be lower. The open windows case was significantly higher in terms of both peak and average ozone (by more than a factor of five) because the air flow rates are high and air flowing in through an open window has no opportunity to be filtered by the envelope. This result for open windows is expected based on the

results of field testing discussed earlier. However, the internal deposition and forced air system filter removal mechanisms are still active and the indoor ozone levels are still much lower than outside. The best ventilation system performance is from the economizer. The economizer ventilates a lot at night when ozone concentrations are low and there mechanical ventilation system is off for ten hours of the day (10 a.m. to 8 p.m.) – thus avoiding ventilation through most of the significant outdoor ozone levels. Also, when operating, the economizer is a very high ventilation rate and can quickly dilute any remaining indoor ozone.

Table 1. Summary of air exchange rates and ozone concentrations.

Climate zone	Ventilation system	Average air exchange rate (h^{-1})	Average indoor ozone level (ppb)	Average indoor:outdoor ratio	Indoor peak ozone level (ppb)
10	continuous exhaust	0.33	2.8	0.053	13.9
	intermittent exhaust	0.33	2.5	0.047	11.0
	HRV	0.45	4.5	0.083	21.8
	CFIS+continuous exhaust	0.39	3.4	0.062	18.3
	continuous supply	0.30	3.7	0.068	14.5
	economizer	1.64	1.8	0.33	11.0
10	open windows	2.49	26	0.467	87
12	continuous exhaust	0.32	2.5	0.050	8.8
	intermittent exhaust	0.32	2.3	0.046	7.3
	HRV	0.44	4.2	0.083	16.2
	CFIS+continuous exhaust	0.38	3.0	0.059	11.4
	continuous supply	0.33	3.7	0.071	9.3
	economizer	1.42	1.4	0.029	7.3

The results show that in each case the indoor ozone concentrations are much lower than those outside, and that ventilation system selection can effect the indoor ozone concentrations by about a factor of three. The peak levels follow the same trends. The two key factors in selecting a ventilation system to minimize ozone levels are: 1. the envelope is good ozone filter, so systems that depressurize the house lead to lower indoor ozone levels, and 2., controls that allow ventilation systems to turn off during high outdoor ozone and ventilate more when outdoor ozone is low.

The above results – in terms of exact ozone concentrations are specific to the locations studies and other locations, with different outdoor ozone profiles would result in changes to indoor concentrations. Similarly the particular changes to indoor:outdoor concentrations will be influenced by home airtightness, weather, and home construction (material interactions with ozone and air flow paths). However, this study was focused on the effects of different ventilation systems, and their relative performance would be preserved for at different locations with changes in ozone, weather and building parameters. Therefore, the conclusions are limited to the relative merits of the ventilation approaches studied here rather than specifics about ozone concentration levels.

10. Conclusions and Recommendations

The following conclusions result from the modeling, simulation and analysis:

- Staying indoors with windows closed significantly (by 80% to 90%) reduces exposure to ozone – even when mechanical ventilation systems are operating due to a combination of envelope filtration and interior deposition.
- Open windows and the resulting unfiltered high ventilation rate significantly reduces the benefits of being indoors.
- Selection of ventilation systems can have a significant impact on indoor:outdoor ozone ratios:
  - Exhaust systems give lower indoor:outdoor ozone ratios (by about 20%) of ozone due to ozone filtration of incoming air in the building envelope.
  - Ventilation systems that can be turned off during high outdoor ozone events and operated at higher air flows at other times (to ensure other pollutants are adequately ventilated) can reduce indoor:outdoor ozone ratios significantly (by about 10-40%)
  - Ventilation systems that have high air flow rates while operating (and cycle off to maintain the same average air flow rate) can produce peak indoor concentrations about double those of other systems

Recommended future activities are:

- Investigate the use of optimized ventilation strategies and air tightening to mitigate indoor ozone
- Further examination of the byproducts of ozone reactions with building materials (such as insulation) and simulating the byproduct formation and indoor levels of byproducts from ozone-initiated chemistry
- Evaluation of the deliberate ozone removal using activated carbon filters integrated into ventilation systems

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Appendix D
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Suggested citation

Moorman JE, Akinbami LJ, Bailey CM, et al. National Surveillance of Asthma: United States, 2001–2010. National Center for Health Statistics. Vital Health Stat 3(35). 2012.

Library of Congress Cataloging-in-Publication Data

National surveillance of asthma: United States, 2001–2010.
p. ; cm. — (Vital and health statistics. Series 3, Analytical and epidemiological studies ; Number 35) (HHS publication ; no. (PHS) 2013–1419)
"November 2012."
Includes bibliographical references.
ISBN 0-8406-0657-5 (alk.paper)
I. National Center for Health Statistics (U.S.) II. Series: Vital & health statistics. Series 3, Analytical and epidemiological studies ; no. 35. III. Series: HHS publication ; no. (PHS) 2013–1419.
[DNLM: 1. Asthma—epidemiology—United States. 2. Asthma—mortality—United States. 3. Health Services—utilization—United States. W2 A N148vc no.35 2012]
614.5'923800973090511—dc23

2012037459

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National Surveillance of Asthma: United States, 2001–2010

U.S. DEPARTMENT OF HEALTH AND HUMAN SERVICES
Centers for Disease Control and Prevention
National Center for Health Statistics

Hyattsville, Maryland
November 2012
DHHS Publication No. (PHS) 2013–1419

National Center for Health Statistics

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Abstract

Background

Asthma is prevalent but treatable: adherence to evidence-based treatment lessens impairment and lowers the risk of future exacerbations.

Objective

This report details recent trends in asthma prevalence, health care use, and mortality since 2001 and presents an overview of trends since 1980.

Methods

Asthma prevalence estimates were obtained from the National Health Interview Survey (2001–2010). Physician office visit data were obtained from the National Ambulatory Medical Care Survey, hospital outpatient department and emergency department (ED) visit data from the National Hospital Ambulatory Medical Care Survey, hospitalization data from the National Hospital Discharge Survey, and death data from the National Vital Statistics System (2001–2009). Two types of rates were calculated: population-based rates based on the total population and risk-based rates based on the population with asthma.

Results

Current asthma prevalence increased from 2001 to 2010. There were no significant changes in rates for hospital outpatient department visits, ED visits, or hospitalizations, whereas risk-based rates for private physician office visits declined. Asthma death rates decreased from 2001 to 2009. Over the long term, asthma prevalence rose more slowly after 2001 than during 1980–1996, asthma hospitalizations declined since 1984 and deaths declined since 1999. Disparities by race and sex for adverse outcomes remained high despite these declines.

Conclusion

Since 2001, asthma prevalence increased, risk-based rates for visits to private physician offices and deaths declined, and risk-based rates for other types of ambulatory visits and for hospitalizations showed no clear trend.

Keywords: prevalence • health care use • morbidity • mortality

National Surveillance of Asthma: United States, 2001–2010

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Introduction

Asthma is a common chronic disorder of the airways characterized by periods of reversible airflow obstruction known as asthma exacerbations or attacks (1,2). Airflow is obstructed by factors which narrow airways in the lungs, including inflammation and airway hyperreactivity (contraction of the small muscles surrounding the airways), in reaction to certain exposures. Exposures may include exercise; airway infections; airborne allergens (e.g., pollen, mold, animal dander, dust mite, and cockroach allergens); occupational exposures (e.g., sensitizing chemicals or dusts); and airborne irritants (e.g., particulate matter and environmental tobacco smoke). Symptoms during an asthma attack may include wheezing, coughing, shortness of breath, and chest tightness or pain. Attacks can range from a nuisance to a life-threatening event. Although many people with asthma have only occasional mild symptoms, others have severe asthma that interferes with daily activity (1). Although causal occupational exposures have been identified, it is not clear how to prevent asthma from developing and there is no cure (1,3). Yet, the means to control and prevent exacerbations in persons who have asthma are well established (1).

Once an exacerbation begins, treatment for short-term relief includes medication to relax the small muscles in the airways and to decrease inflammation. The cornerstone of treatment, however, is long-term prevention to control inflammation and airway hyperreactivity (1). Asthma management strategies include adequate patient education, consistent use of preventive medication, and control of environmental factors that affect asthma (home-based, multi-trigger multi-component interventions). These strategies have been shown to be effective in improving the health of individuals affected by asthma and are integral to reducing the impact of asthma in the United States (1,2,4).

Public health goals for asthma, outlined in the Healthy People objectives, focus on mitigating the impact of asthma rather than on reducing asthma incidence or prevalence (5). Public health efforts, therefore, include interventions to reduce asthma deaths, hospitalizations, and emergency department (ED) visits, and to increase the proportion of persons who receive health care consistent with the Guidelines for the Diagnosis and Management of Asthma (1). This report is the fourth in a series of national asthma surveillance summaries (2,7,8). It presents data from national health surveys conducted by the Centers for

Disease Control and Prevention's (CDC) National Center for Health Statistics (NCHS), and describes the overall asthma surveillance picture for the United States since 2001. Two types of rates for asthma outcomes are presented: population-based rates that indicate the rate for the entire U.S. population and population subgroups (e.g., race and ethnic group, sex, and geographic region) and reflect the overall burden on the general population; and risk-based rates for which the denominator is restricted to estimates of the number of persons who have asthma and are at risk for asthma-related outcomes. Risk-based measures control for differences in asthma prevalence among population subgroups and across time so that the magnitude of the outcome and the trends among people with asthma can be seen more clearly. The measures and definitions used are consistent with those presented in the 2007 surveillance summary so that the historic estimates in that previous report can be directly compared with estimates in this report (2). An overview of long-term trends from 1980 for asthma prevalence, health care use, and mortality is also presented.

Methods

Data Sources

Asthma prevalence and morbidity data are obtained from national surveys conducted by NCHS. Asthma prevalence data are self-reported by respondents to the National Health Interview Survey (NHIS) 2001–2010; asthma office visit data are obtained from the National Ambulatory Medical Care Survey (NAMCS) 2001–2009; hospital outpatient department visits and ED data are obtained from the National Hospital Ambulatory Medical Care Survey (NHAMCS) 2001–2009; asthma hospitalization data are obtained from the National Hospital Discharge Survey (NHDS) 2001–2009; and asthma death data are obtained from the National Vital Statistics System (NVSS), Underlying Cause of Death files 2001–2009. Details on each data source can be found in Appendix I: Data Sources and Limitations.

Analytic Methods

For all survey data, survey weights were used to calculate population-based estimates. With the exception of death record data from 1980–2009 and hospitalization data from 1980–1987, standard errors were calculated using SUDAAN software to account for each survey's complex sample design (9). Standard errors for death rates were based on the number of deaths (10). For hospitalizations between 1980 and 1987, standard errors were estimated using relative standard error tables provided by NCHS' Division of Health Care Statistics (11). Reliability of estimates was based on the relative standard error (RSE), which is calculated by dividing the estimate into its standard error and multiplying by 100. Estimates with an RSE between 30% and 50% are noted. Estimates with an RSE greater than 50% are not shown. For mortality data, estimates based on fewer than 20 deaths are also not shown.

Significance testing for trends was performed with Joinpoint software (12). Joinpoint regression characterizes trends as joined linear segments on a logarithmic scale. A joinpoint is the year when two segments with different slopes meet. Average annual percentage change for each segment was estimated based on the natural logarithm of rates. A 95% confidence interval around the annual percent change was used to determine whether the annual percent change for each segment is significantly different from zero.

Data on demographic characteristics (age, sex, race and ethnicity, and geographic region) were standardized across all data sources to the extent possible. Poverty status and detailed race and ethnicity groups were only available for prevalence data (NHIS). Age groups 15–19, 20–24, and 25–34 are not shown separately for hospital outpatient (NHAMCS) and office visits (NAMCS) due to small sample sizes. Hispanic ethnicity was not available for hospital discharge data (NHDS).

To obtain sufficient sample sizes to produce stable estimates for demographic subgroups, 3 years of data for the most recent period were combined and presented as 3-year average annual estimates. For NHIS, data are presented for 2008–2010, and for the remaining data sources, data are presented for 2007–2009. Risk-based rates were calculated by dividing estimates of the number of events (office visits, hospital outpatient visits, ED visits, and hospitalizations) and the number of deaths by the estimated number of persons with current asthma from NHIS for the same demographic group and the same year or the same 3-year period. Population-based rates were calculated by dividing estimates of the number of events (office visits, hospital outpatient visits, ED visits, and hospitalizations) and the number of deaths by the estimated population size for the same demographic group and the same year or the same 3-year period.

Age-adjusted estimates are provided only for population-based rates to account for age-related differences over time and between population subgroups. Risk-based rates are not age adjusted because they already account for asthma prevalence differences. Comparisons can be made between groups without concern that variations in asthma prevalence are contributing to observed differences. Additionally, there were no substantial changes in the age distribution among the population with current asthma or with at least one asthma attack in the previous year.

Additional details on methods can be found in Appendix II: Technical Notes. Additional details on each demographic characteristic and each asthma-specific measure can be found in Appendix III: Definition of Terms.

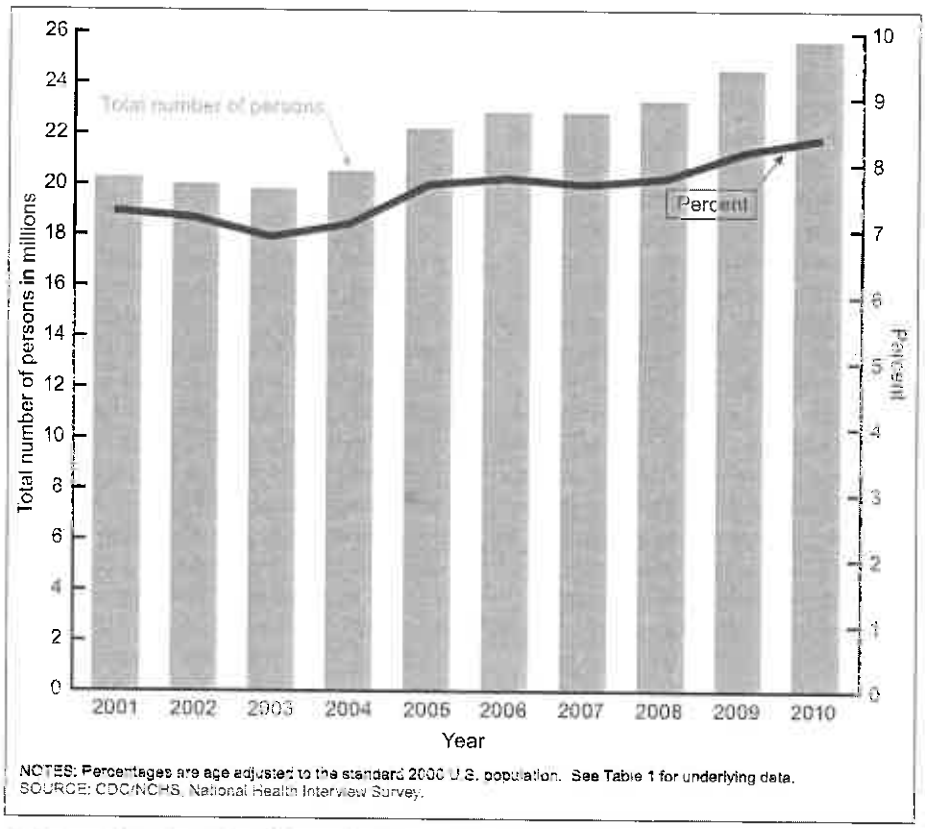
Current Asthma Prevalence and Asthma Attack Prevalence

Estimates of asthma prevalence indicate the number of persons and percentage of the population with asthma at a given point in time. Current asthma prevalence estimates, available from 2001 through 2010, indicate the number and percentage of people who reported receiving a diagnosis of asthma and still having asthma at the time of the health survey. Asthma attack prevalence estimates, available from 1997 through 2010, indicate the number and percentage of people with an asthma diagnosis who reported having at least one asthma attack in the previous 12 months. This measure is an indicator of suboptimal asthma control. Those who report experiencing an attack are at risk for additional adverse outcomes such as missing work or school, or requiring acute medical treatment or hospitalization.

This section analyzes trends for 2001–2010 for current asthma prevalence and asthma attack prevalence for population-based rates, risk-based rates, and counts of persons with current asthma or at least one asthma attack within the past year. Differences between population subgroups are assessed over this time period, and increased demographic detail using average annual estimates is presented for the period 2008–2010. Additional details not included in the figures can be found in Tables 1–6.

Current Asthma Prevalence Trends

Number of persons with asthma—The number of persons with asthma increased 2.9% each year, from 20.3 million persons in 2001 to 25.7 million persons in 2010 (Figure 1). Of the 25.7 million, 7.0 million were children and 18.7 million were adults. Among adults, 3.1 million people aged 65 and over had asthma. By race, 19.1 million were white, 4.7 million were black, and 1.9 million were of other races. By ethnicity, 3.6 million were Hispanic and 22.1 million were non-Hispanic.



NOTES: Percentages are age adjusted to the standard 2000 U.S. population. See Table 1 for underlying data.
SOURCE: CDC/NCHS, National Health Interview Survey.

Figure 1. Current asthma prevalence: United States, 2001–2010

Trends in current asthma prevalence overall, by age group and sex—Current asthma prevalence increased at a rate of 1.5% per year to a prevalence of 8.4% in 2010 (Figure 1). Among children aged 0–17 years, current asthma prevalence increased at a rate of 1.4% per year, and among adults aged 18 and over, prevalence increased at a rate of 2.1% per year. Current asthma prevalence increased at a rate of 1.8% per year among males and females (Table 1).

Trends in current asthma prevalence, by race and ethnicity—Among white persons, current asthma prevalence increased at a rate of 1.4% per year to a prevalence of 7.8% in 2010. Among black persons, current asthma prevalence increased at a rate of 3.2% per year to 11.9% in 2010. By ethnicity, current asthma prevalence rose among Hispanic persons at a rate of 3.2% per year to 7.2% in 2010. Among non-Hispanic persons, prevalence rose at a rate of 1.9% per year to 8.7% in 2010 (Table 1).

Current Asthma Prevalence by Selected Characteristics

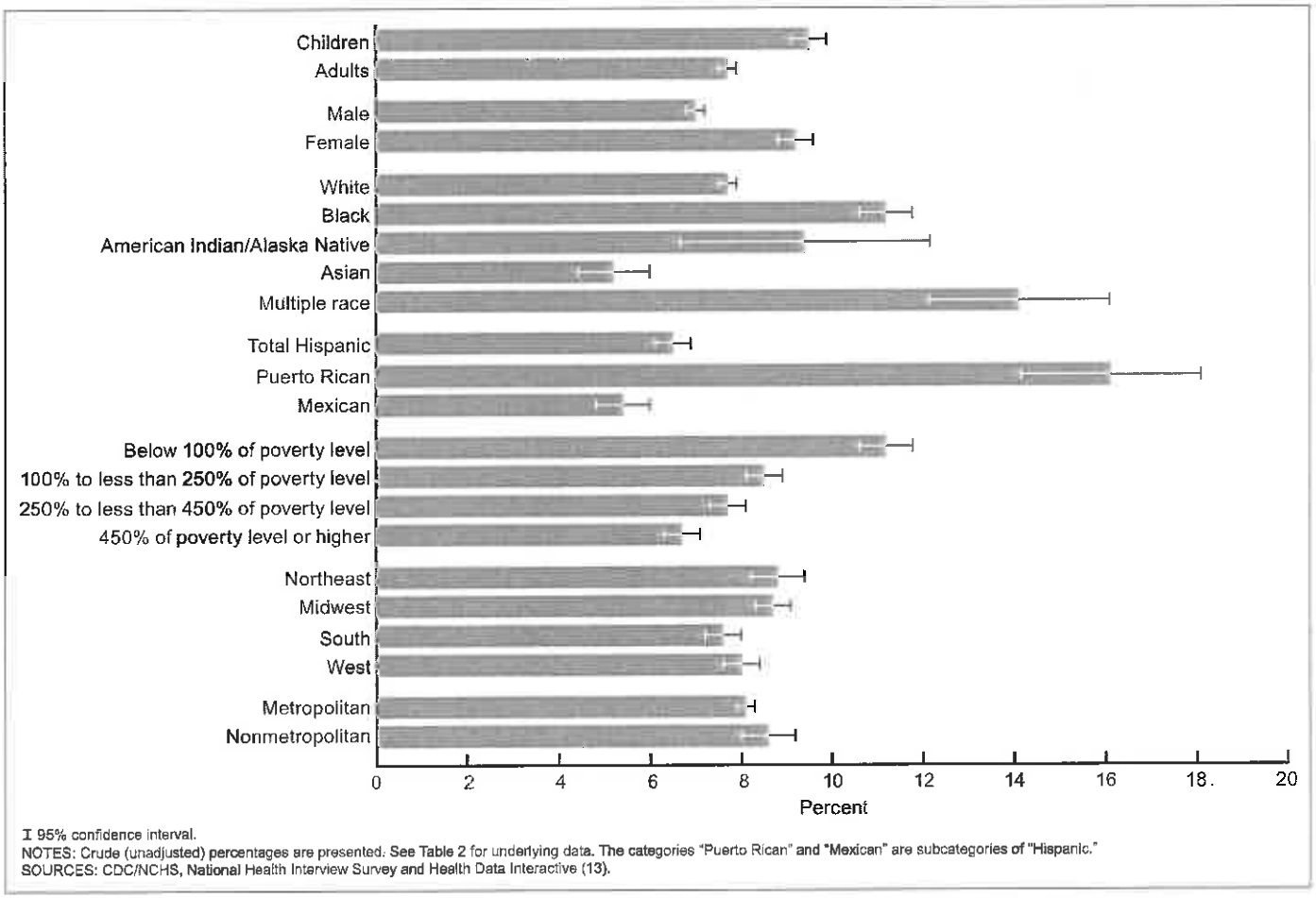
Age group and sex differences in 2008–2010 current asthma prevalence—For the period 2008–2010, average annual current asthma prevalence was higher in children aged 0–17 years (9.5%) than in adults aged 18 and over (7.7%), and was higher in females (9.2%) than in males (7.0%) (Figure 2). However, the difference in prevalence between males and females varied by age group. Among children, boys had higher prevalence than girls (11.1% compared with 7.8%) whereas among adults, men had lower prevalence than women (5.7% compared with 9.7%) (Table 2).

Race differences in 2008–2010 current asthma prevalence—Current asthma prevalence was higher in black persons (11.2%) than in white persons (7.7%) (Figure 2, Table 2).

Ethnicity differences in 2008–2010 current asthma prevalence—Among Hispanic persons, Puerto Rican persons had higher prevalence (16.1%) compared with Mexican persons (5.4%) (Figure 2, Table 2).

Differences by poverty status in 2008–2010 current asthma prevalence—Current asthma was more prevalent among persons with family income below 100% of the federal poverty threshold (11.2%) than among persons with family income at or above the federal poverty threshold (8.5% for 100% to less than 250% of the poverty threshold, 7.8% for 250% to less than 450% of the poverty threshold, and 6.7% for at or above 450% of the poverty threshold). Asthma prevalence was significantly lower for each successively higher poverty level group (Figure 2, Table 2).

Differences by geographic region and urbanicity in 2008–2010 current asthma prevalence—The current asthma prevalence rate was higher in the Northeast (8.8%) than in the South (7.6%) or in the West (8.0%), and was higher in the Midwest (8.7%) than in the South (7.6%). Current asthma prevalence did not differ between metropolitan and nonmetropolitan areas (Figure 2, Table 2).



I 95% confidence interval.
NOTES: Crude (unadjusted) percentages are presented. See Table 2 for underlying data. The categories "Puerto Rican" and "Mexican" are subcategories of "Hispanic."
SOURCES: CDC/NCHS, National Health Interview Survey and Health Data Interactive (13).

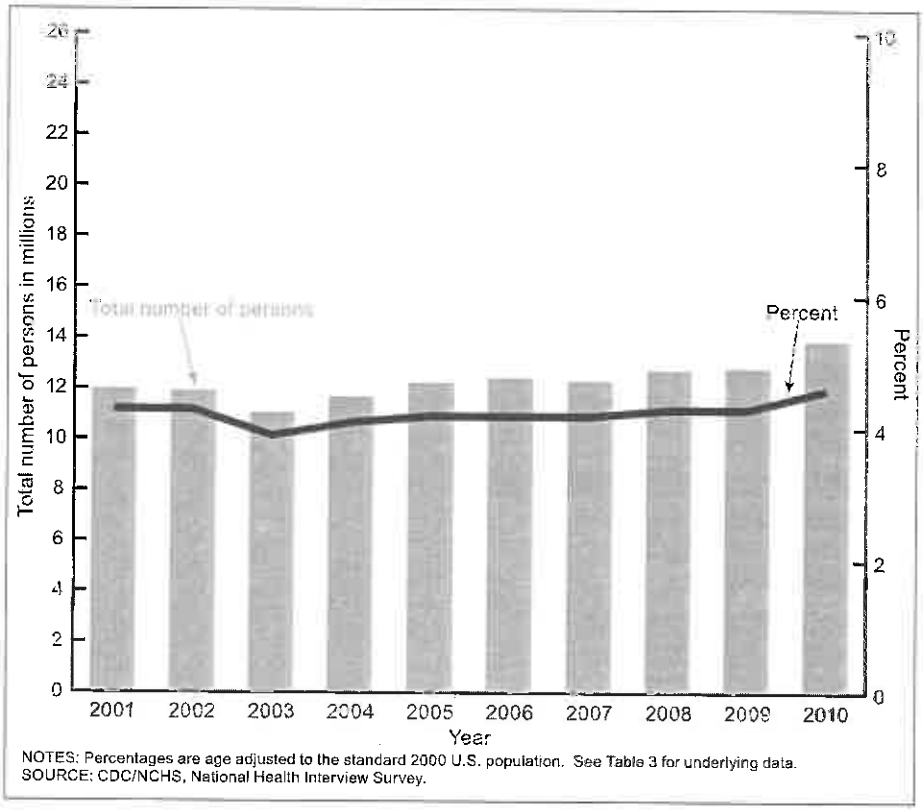
Figure 2. Current asthma prevalence, by age group, sex, race and ethnicity, poverty status, geographic region, and urbanicity: United States, average annual 2008–2010

Asthma Attack Prevalence Trends

Number of persons with an asthma attack in the previous 12 months—The number of persons with at least one asthma attack in the previous 12 months was stable from 2001 to 2003 then increased 2.6% per year from 11.0 million in 2003 to 13.9 million in 2010 (Figure 3). Among this number, 4.3 million were children and 9.6 million were adults; 5.4 million were males and 8.5 million were females. By race, 10.4 million were white, 2.3 million were black, and 1.2 million were of another race. By ethnicity, 2.0 million were Hispanic and 11.9 million were non-Hispanic.

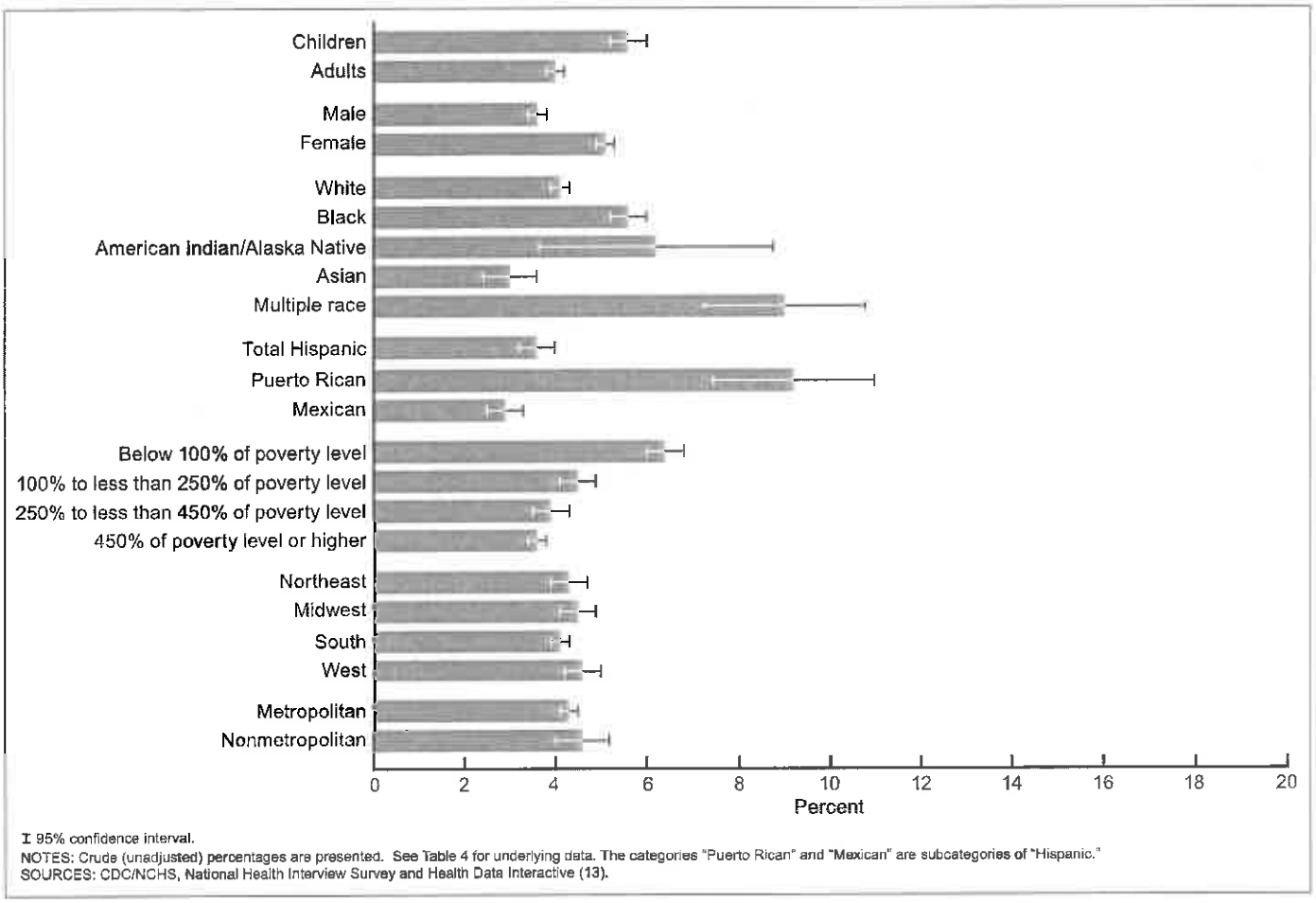
Trends in population-based asthma attack prevalence overall, by age group and sex—The percentage of people with at least one asthma attack in the previous 12 months remained level during the period from 2001 to 2003 (range: 3.9% to 4.3%) but increased 1.6% per year from 2003 to 2010, from 3.9% to 4.6% (Figure 3, Table 3). Children had asthma attack prevalence rates (range: 5.2% to 5.8%) that were higher than adult rates (range: 3.3% to 4.2%). Females had higher asthma attack prevalence (range: 4.4% to 5.4%) compared with males (range: 3.1% to 3.7%) (Table 3).

Trends in population-based asthma attack prevalence, by race and ethnicity—Black persons had higher asthma attack prevalence (range: 4.6% to 5.8%) compared with white persons (range: 3.7% to 4.3%) in most years during the period. In 2005 and 2006 there was no significant difference between black and white persons. For race and Hispanic groups except white persons, asthma attack prevalence rates remained level for the period. For white persons, asthma attack prevalence increased 1.6% per year from 2003 to 2010 (Table 3).



NOTES: Percentages are age adjusted to the standard 2000 U.S. population. See Table 3 for underlying data.
SOURCE: CDC/NCHS, National Health Interview Survey.

Figure 3. Asthma attack prevalence (population-based): United States, 2001–2010



I 95% confidence interval.
NOTES: Crude (unadjusted) percentages are presented. See Table 4 for underlying data. The categories "Puerto Rican" and "Mexican" are subcategories of "Hispanic."
SOURCES: CDC/NCHS, National Health Interview Survey and Health Data Interactive (13).

Figure 4. Asthma attack prevalence (population-based), by age group, sex, race and ethnicity, poverty status, geographic region, and urbanicity: United States, average annual 2008–2010

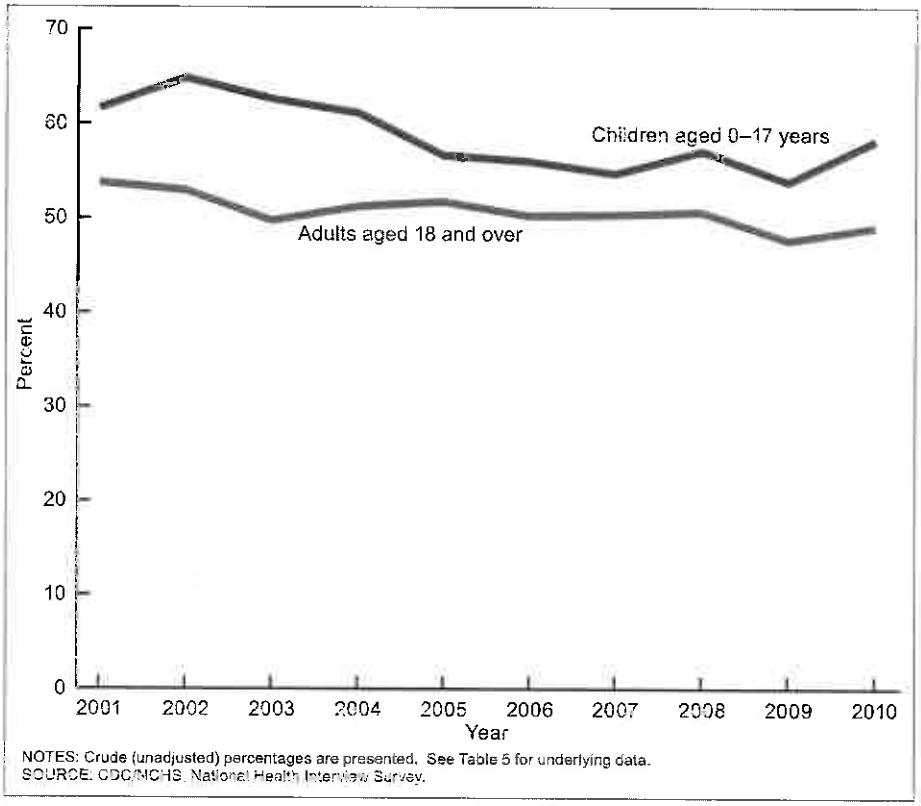
Asthma Attack Prevalence by Selected Characteristics

Asthma attack prevalence 2008–2010— Differences by age, gender, race and ethnicity, and poverty for population-based asthma attack rates were similar to those found for current asthma prevalence (Figure 2). Children had higher rates than adults, females had higher rates than males, black persons had higher rates than white persons. persons of Puerto Rican descent had higher rates than those of Mexican descent, and those below the poverty level had higher rates than those above the poverty level. However, unlike current asthma prevalence, population-based asthma attack prevalence rates for the South, the Northeast, and the Midwest were similar (Figure 4, Table 4).

Asthma Attack Prevalence Trends Among Children and Adults

Trends in asthma attack prevalence among persons with current asthma— Asthma attack prevalence among persons with current asthma decreased from 55.8% in 2001 to 51.9% in 2010, at a rate of 1.0% per year. This decrease in the proportion reporting an attack in the past 12 months represents progress in asthma management (Figure 5, Table 5). However, Figure 3 shows an increasing trend in asthma attack prevalence among the general population from 2003 to 2010 (population-based rate). The population-based asthma attack prevalence rate increased because current asthma prevalence increased (see Figure 1). That is, a larger proportion of the population had asthma and was at risk for an asthma attack, which outweighed the lower risk among people with asthma of having an asthma attack. The result was that for asthma attack prevalence, the population-based rate increased even while the risk-based rate declined.

***Risk-based asthma attack prevalence among children*—**In 2001, 61.7% of children with current asthma had at least one asthma attack in the previous 12 months compared with 58.3% in 2010; the rate of decrease per year from 2001 to 2010 was 1.5% (Table 5). In each year from 2001 to 2010, a higher percentage of children with current asthma had at least one attack in the previous 12 months compared with adults with current asthma.



NOTES: Crude (unadjusted) percentages are presented. See Table 5 for underlying data.
SOURCE: CDC/NCHS, National Health Interview Survey.

Figure 5. Asthma attack prevalence among children and adults with current asthma (risk-based): United States, 2001–2010

***Risk-based asthma attack prevalence among adults*—**In 2001, 53.8% of adults with asthma had at least one asthma attack in the previous 12 months compared with 49.1% in 2010. Asthma attack prevalence among adults with current asthma declined from 2001 to 2010 at a rate of 1.0% per year (Table 5).

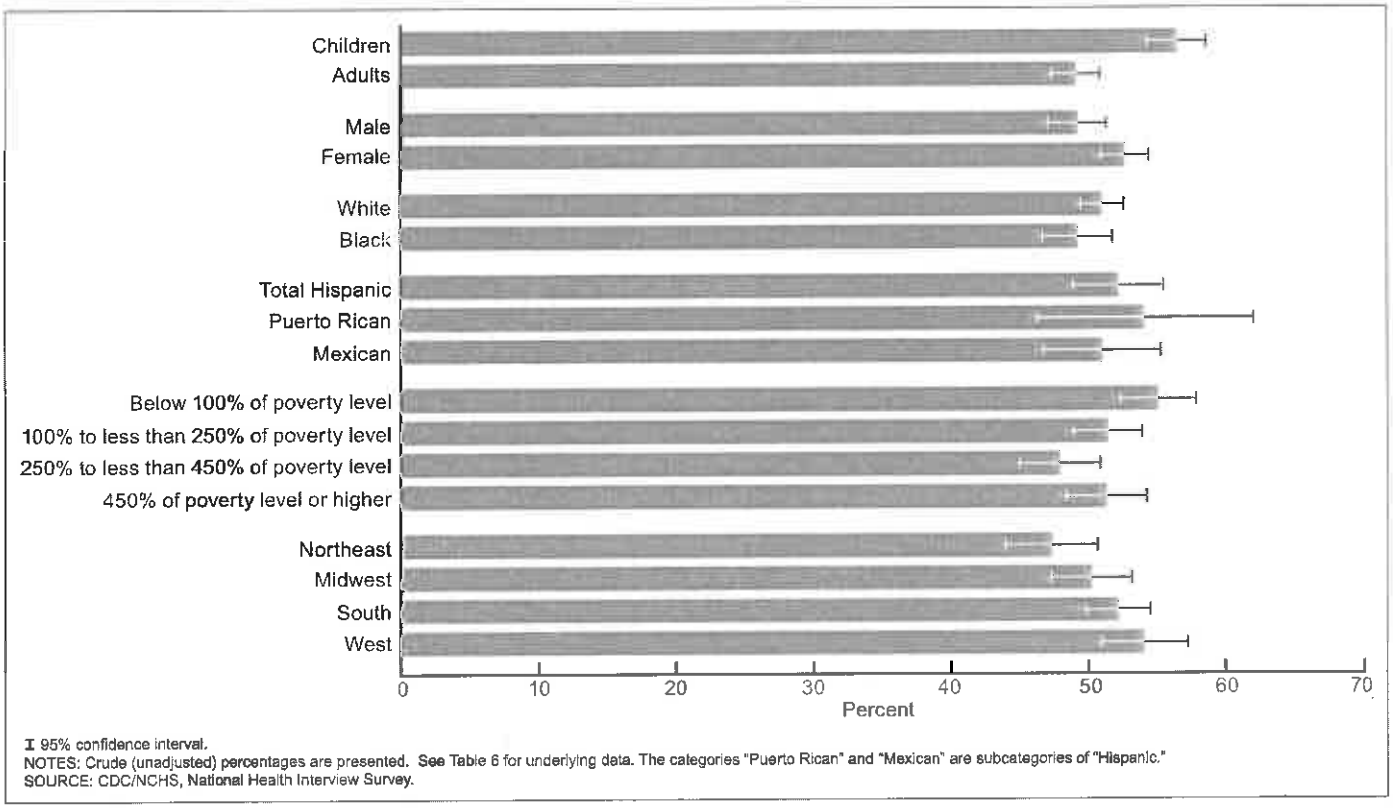
Asthma Attack Prevalence Among Persons With Current Asthma

Age group differences in 2008–2010 asthma attack prevalence among persons with current asthma—Overall, an average annual 51.2% of persons with current asthma reported having at least one asthma attack during the previous 12 months. Children aged 0–17 years with current asthma were more likely to have had at least one attack during the previous 12 months (56.5%) than adults aged 18 and over (49.1%) (Figure 6, Table 6).

Differences by sex and race and ethnicity in 2008–2010 asthma attack prevalence among persons with current asthma—Females with current asthma were more likely to have had an attack during the previous 12 months (52.7%) than males (49.2%). Asthma attack prevalence among persons with asthma did not vary significantly by race or ethnicity (Figure 6, Table 6).

Differences by poverty status in 2008–2010 asthma attack prevalence among persons with current asthma—Among persons with current asthma, having at least one asthma attack in the previous 12 months was more prevalent among those with family income less than 100% of the federal poverty threshold (55.1%) than among persons with family income between 250% and less than 450% of the poverty threshold (47.9%) (Figure 6, Table 6).

Differences by geographic region in 2008–2010 asthma attack prevalence among persons with current asthma—The risk-based asthma attack prevalence rate was higher in the South (52.2%) and the West (54.1%) than in the Northeast (47.3%) (Figure 6, Table 6). This regional difference is the reverse of that seen for asthma prevalence. Current asthma prevalence was lower in the South and the West than in the Northeast (Figure 2, Table 2).



I 95% confidence interval.
NOTES: Crude (unadjusted) percentages are presented. See Table 6 for underlying data. The categories "Puerto Rican" and "Mexican" are subcategories of "Hispanic."
SOURCE: CDC/NCHS, National Health Interview Survey.

Figure 6. Asthma attack prevalence among persons with current asthma (risk-based), by age group, sex, race and ethnicity, poverty status, and geographic region: United States, average annual 2008–2010

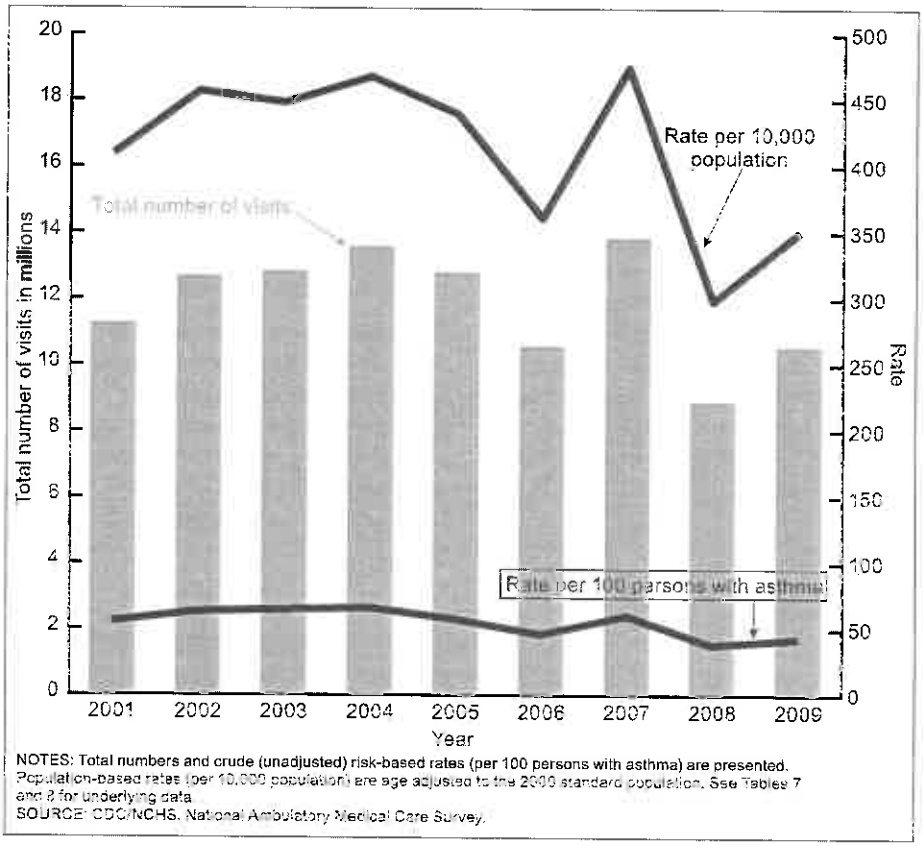
Physician Office Visits and Hospital Outpatient Department Visits

Estimates for asthma physician office and hospital outpatient visits include the number of visits, the rate of visits for the general population (population-based rates), and the rate of visits for the population with current asthma (risk-based rates). Routine visits to a physician office or hospital outpatient clinic for preventive asthma care is a key component of asthma management and maintaining good asthma control (1). However, ambulatory care estimates based on visits to physician offices or hospital outpatient departments also include visits for patients with exacerbations of their asthma. To the extent that worsening symptoms are reversed and asthma control is improved, ambulatory visits in these nonacute settings may avert more severe outcomes such as asthma ED visits or hospitalizations.

This section presents 2001–2009 trends for asthma visits to private physician offices and hospital outpatient departments for population-based rates, risk-based rates, and counts of visits. Differences between detailed demographic subgroups are assessed using average annual estimates for the period 2007–2009. Additional detail not included in the figures can be found in Tables 7–12.

Physician Office Visit Trends

Number of physician office visits—There was fluctuation in the number of asthma physician office visits from 2001 to 2009, but no clear trend. In 2009, there were 10.6 million asthma physician office visits (Figure 7).



NOTES: Total numbers and crude (unadjusted) risk-based rates (per 100 persons with asthma) are presented. Population-based rates (per 10,000 population) are age adjusted to the 2000 standard population. See Tables 7 and 8 for underlying data.
SOURCE: CDC/NCHS, National Ambulatory Medical Care Survey.

Figure 7. Asthma physician office visits: United States, 2001–2009

Population-based rate of physician office visits—The population-based rate of physician office visits for asthma fluctuated but did not follow a significant trend from 2001 to 2009. In 2009, there were 351.2 visits per 10,000 population (Figure 7, Table 7).

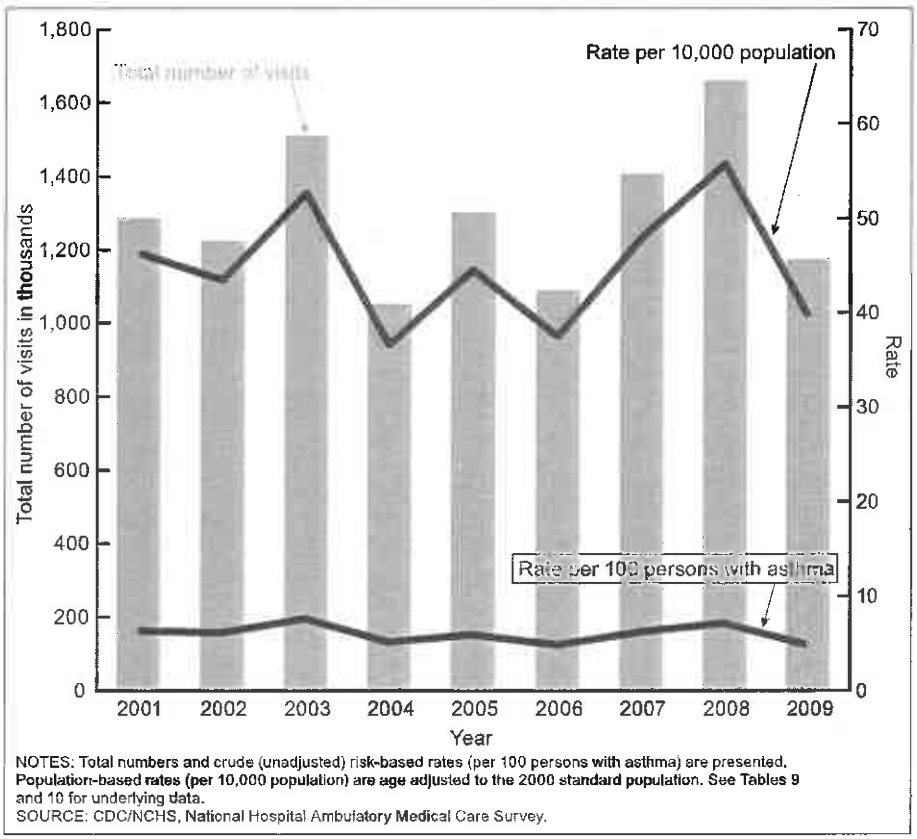
Risk-based rate of physician office visits—From 2001 to 2009, there was a 5.5% decrease per year in the risk-based rate of physician office visits for asthma. In 2009, there were 43.0 visits per 100 persons with asthma (Figure 7, Table 8).

Hospital Outpatient Department Visit Trends

Number of hospital outpatient department visits—There was no significant trend in the number of asthma hospital outpatient department visits from 2001 to 2009. In 2009, there were 1.2 million asthma hospital outpatient department visits (Figure 8).

Population-based rate of hospital outpatient department visits—The population-based rate of asthma hospital outpatient department visits for asthma fluctuated but did not follow a significant trend from 2001 to 2009. In 2009, there were 39.6 visits per 10,000 population (Figure 8, Table 9).

Risk-based rate of hospital outpatient department visits—From 2001 to 2009, there was no significant trend in the risk-based rate of hospital outpatient department visits for asthma. In 2009, there were 4.8 visits per 100 persons with asthma (Figure 8, Table 10).



NOTES: Total numbers and crude (unadjusted) risk-based rates (per 100 persons with asthma) are presented. Population-based rates (per 10,000 population) are age adjusted to the 2000 standard population. See Tables 9 and 10 for underlying data.
SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

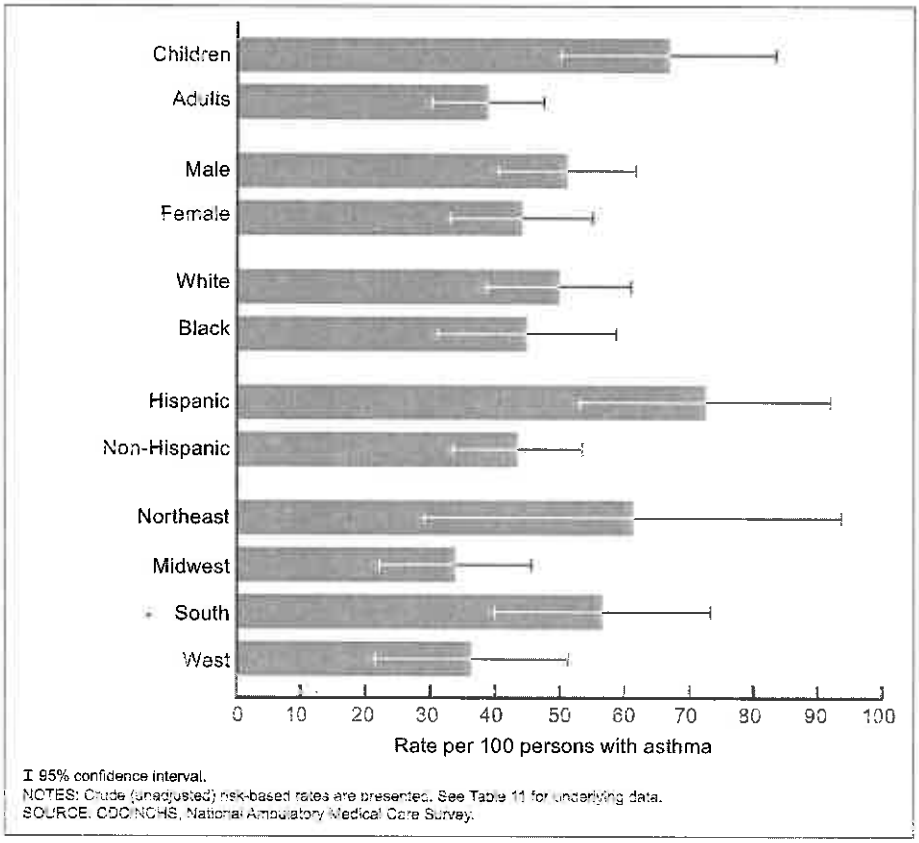
Figure 8. Asthma hospital outpatient department visits: United States, 2001–2009

Physician Office Visits by Selected Characteristics

Differences in risk-based rates of physician office visits, by age group and sex—For the period 2007–2009, the average annual rate of physician office visits for asthma was higher for children aged 0–17 years compared with adults aged 18 and over (66.9 compared with 38.9 visits per 100 persons with asthma). Males and females had similar rates (Figure 9, Table 11).

Differences in risk-based rates of physician office visits, by race and ethnicity—For the period 2007–2009, the average annual rate of physician office visits for asthma for white and black persons was similar. The rate for Hispanic persons was higher than the rate for non-Hispanic persons (72.6 compared with 43.5 per 100 persons with asthma) (Figure 9, Table 11).

Differences in risk-based rates of physician office visits, by geographic region—The average annual rate of physician office visits for asthma for persons living in the South (56.6 visits per 100 persons with asthma) was higher than for those living in the Midwest (33.9 per 100 persons with asthma) (Figure 9, Table 11).



I 95% confidence interval.
NOTES: Crude (unadjusted) risk-based rates are presented. See Table 11 for underlying data.
SOURCE: CDC/NCHS, National Ambulatory Medical Care Survey.

Figure 9. Asthma physician office visit rates (risk-based), by age group, sex, race and ethnicity, and geographic region: United States, average annual 2007–2009

Hospital Outpatient Department Visits by Selected Characteristics

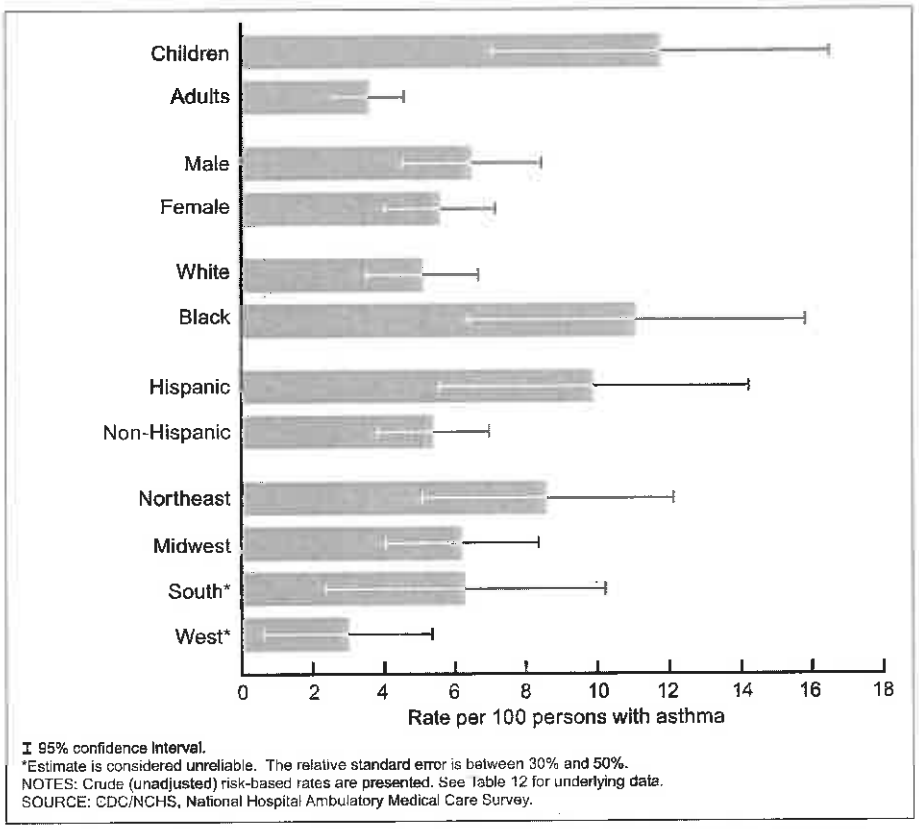
Differences in risk-based rates of hospital outpatient department visits, by age group and sex—For the period 2007–2009, the average annual rate of hospital outpatient department visits for asthma was higher for children aged 0–17 years compared with adults aged 18 and over (11.8 compared with 3.6 visits per 100 persons with asthma). There was no difference by sex (Figure 10, Table 12).

Differences in risk-based rates of hospital outpatient department visits, by race and ethnicity—The average annual rate of hospital outpatient department visits for asthma was lower for white persons (5.1 visits per 100 persons with asthma) compared with black persons (11.1 visits per 100 persons with asthma). Compared with non-Hispanic persons with current asthma, Hispanic persons had a higher average annual rate of hospital outpatient department visits for asthma (9.9 compared with 5.4 per 100 persons with asthma) (Figure 10, Table 12).

Differences in risk-based rates of hospital outpatient department visits, by geographic region—There were no significant differences in risk-based rates of hospital outpatient department visits by geographic region (Figure 10, Table 12).

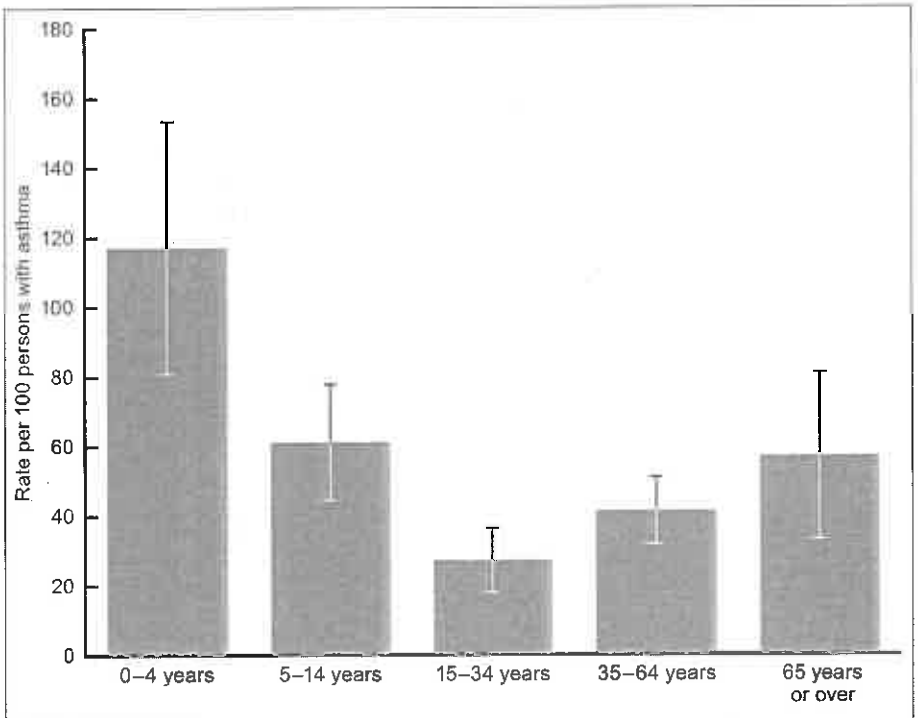
Physician Office Visits by Age Group

Differences in risk-based rates of physician office visits, by detailed age group—The average annual asthma physician office visit rate for children aged 0–4 years was the highest at 117.2 per 100 persons with asthma. For persons aged 15–34, the rate was the lowest at 27.2 per 100 persons with asthma (Figure 11, Table 11).



I 95% confidence interval.
*Estimate is considered unreliable. The relative standard error is between 30% and 50%.
NOTES: Crude (unadjusted) risk-based rates are presented. See Table 12 for underlying data.
SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

Figure 10. Asthma hospital outpatient department visit rates (risk-based), by age group, sex, race and ethnicity, and geographic region: United States, average annual 2007–2009



I 95% confidence interval.
NOTES: Crude (unadjusted) risk-based rates are presented. See Table 11 for underlying data.
SOURCE: CDC/NCHS, National Ambulatory Medical Care Survey.

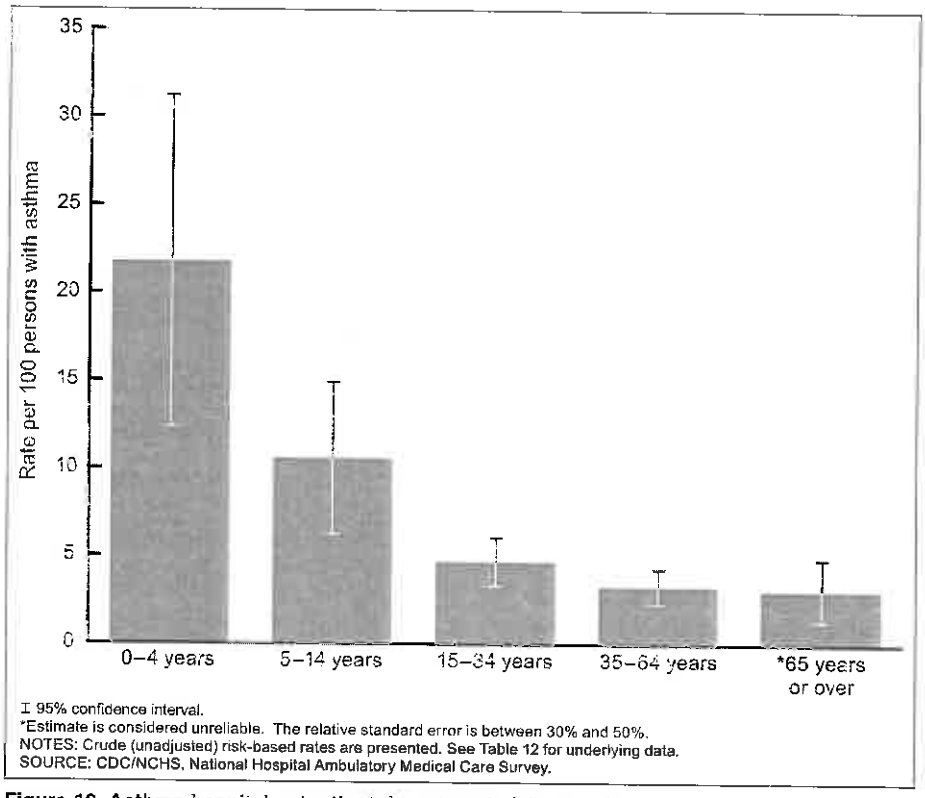
Figure 11. Asthma physician office visit rates (risk-based), by detailed age group: United States, average annual 2007–2009

Hospital Outpatient Department Visits by Age Group

Differences in risk-based rates of hospital outpatient department visits, by detailed age group—Average annual asthma hospital outpatient department visit rates for children aged 0–4 years were the highest at 21.8 per 100 persons with asthma. Visit rates were also higher for children aged 5–14 years (10.6) compared with older age groups (4.7 for those aged 15–34, 3.3 for those aged 35–64, and 3.1 for those aged 65 or over) (Figure 12, Table 12).

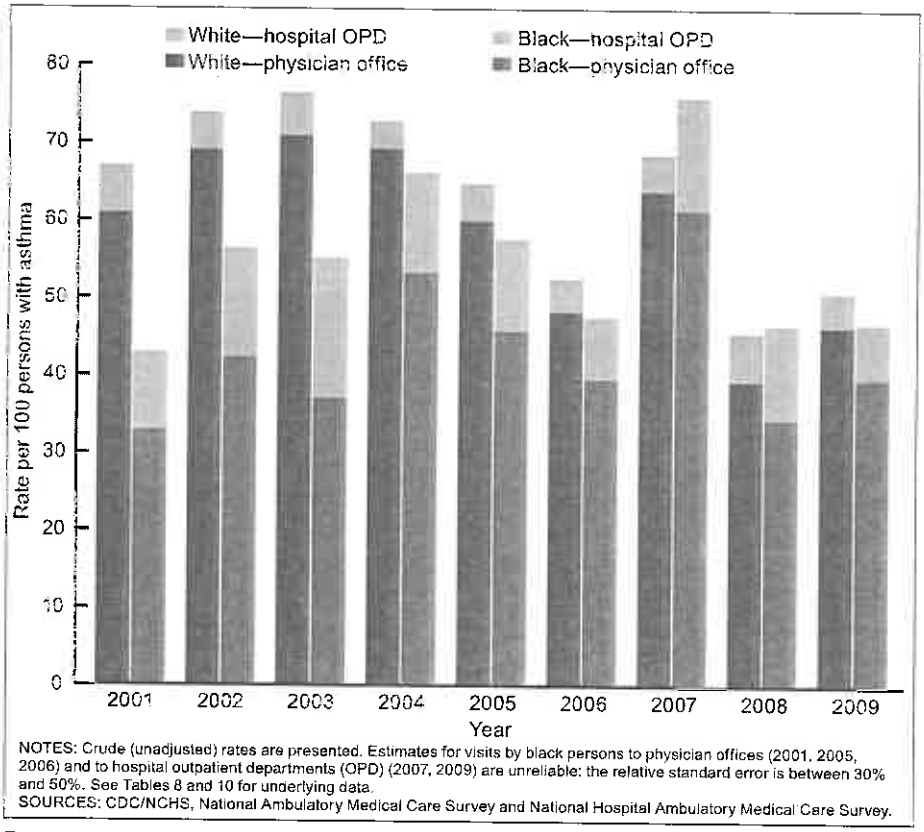
Physician Office and Hospital Outpatient Department Visit Trends by Race

Trends for health care encounter rates for persons with asthma in primary care settings (risk-based), by race—At the beginning of the period 2001–2009, black persons had lower visit rates for primary health care for asthma. Given their higher rates of adverse outcomes relative to white persons, the lower primary visit rates among black persons could indicate underutilization of preventive services. However, encounters in primary care settings increased for black persons with asthma relative to white persons so that during the latter part of the period, total primary care encounter rates were similar by race (Figure 13).



Ŧ 95% confidence interval.
*Estimate is considered unreliable. The relative standard error is between 30% and 50%.
NOTES: Crude (unadjusted) risk-based rates are presented. See Table 12 for underlying data.
SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

Figure 12. Asthma hospital outpatient department visit rates (risk-based), by detailed age group: United States, average annual 2007–2009



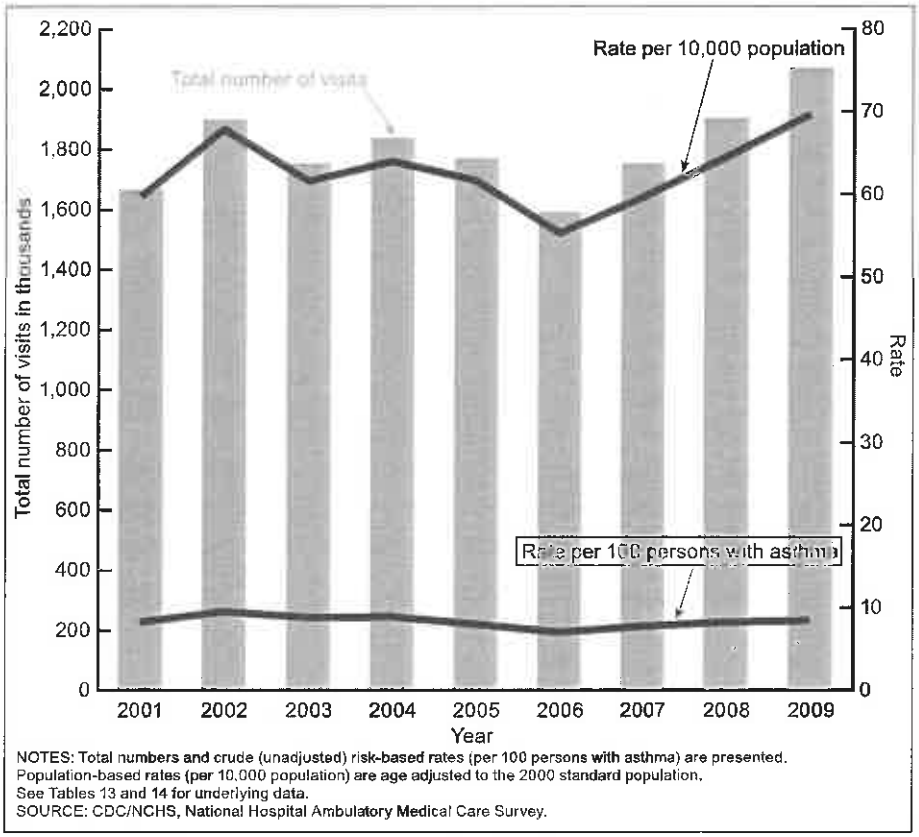
NOTES: Crude (unadjusted) rates are presented. Estimates for visits by black persons to physician offices (2001, 2005, 2006) and to hospital outpatient departments (OPD) (2007, 2009) are unreliable: the relative standard error is between 30% and 50%. See Tables 8 and 10 for underlying data.
SOURCES: CDC/NCHS, National Ambulatory Medical Care Survey and National Hospital Ambulatory Medical Care Survey.

Figure 13. Asthma physician office visit rates and hospital outpatient department visit rates (risk-based), by race: United States, 2001–2009

Emergency Department Visits

Asthma ED measures include the count of visits, the rate of visits for the general population (population-based rates,) and the rate for the population with current asthma (risk-based rates). Visiting the ED for an asthma exacerbation is a key indicator of poorly controlled asthma and of risk for future asthma exacerbations (1). There are specific recommendations for patient education to help prevent future ED visits, including focused and targeted patient education in the physician office and ED setting (assessing inhaler technique, instructions for medication, and steps to follow for worsening symptoms) and referral for follow-up asthma care (1).

This section presents 2001–2009 trends for asthma ED visits for population-based rates, risk-based rates, and counts of visits. Differences between population subgroups are assessed over this period, and more detailed demographic groups are compared using average annual estimates for the period 2007–2009. Additional detail not provided in the figures can be found in Tables 13–15.



NOTES: Total numbers and crude (unadjusted) risk-based rates (per 100 persons with asthma) are presented. Population-based rates (per 10,000 population) are age adjusted to the 2000 standard population. See Tables 13 and 14 for underlying data.
SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

Figure 14. Asthma emergency department visits: United States, 2001–2009

ED Visit Trends

Number of asthma ED visits—Over the period 2001 through 2009, the number of ED visits fluctuated without a clear trend. In 2009, there were 2.1 million asthma ED visits (Figure 14).

Population-based rates of asthma ED visits—The ED visit rate per 10,000 population fluctuated but did not follow a consistent trend. In 2009, there were 69.7 ED visits per 10,000 population (Figure 14, Table 13).

Risk-based rates of asthma ED visits—There was also no significant trend in the risk-based based rates of ED visits. In 2009, there were 8.4 ED visits per 100 persons with asthma; a similar rate persisted throughout the period (Figure 14, Table 14).

ED Visits by Selected Characteristics

Differences in risk-based rates of ED visits, by age group and sex—For the period 2007–2009, children aged 0–17 years had a higher ED visit rate compared with adults aged 18 and over (10.7 compared with 7.0 per 100 persons with asthma). There was no difference in ED rates by sex (8.7 and 7.6 per 100 persons with asthma for males and females, respectively) (Figure 15, Table 15).

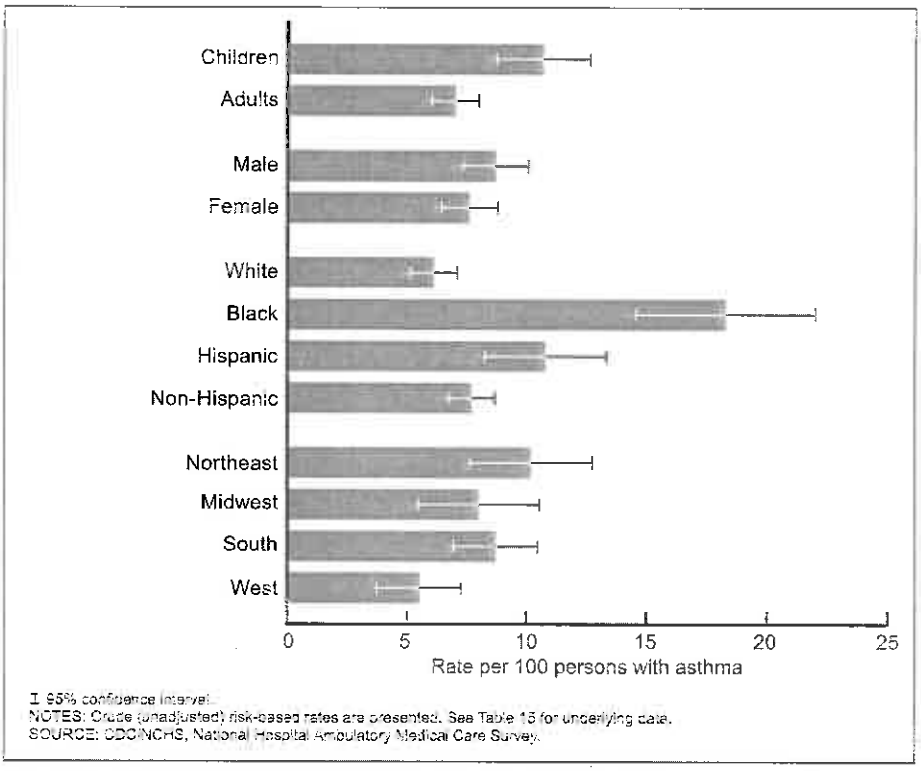
Differences in risk-based rates of ED visits, by race and ethnicity—Black persons had an ED visit rate three times as high as white persons (18.3 compared with 6.1 per 100 persons with asthma). Hispanic persons had a higher ED visit rate than non-Hispanic persons (10.8 compared with 7.7 per 100 persons with asthma) (Figure 15, Table 15).

Differences in risk-based rates of ED visits, by geographic region—Among geographic regions, the ED visit rate was higher in the Northeast (10.2 per 100 persons with asthma) and in the South (8.7) than in the West (5.5) (Figure 15, Table 15).

ED Visit Trends by Race

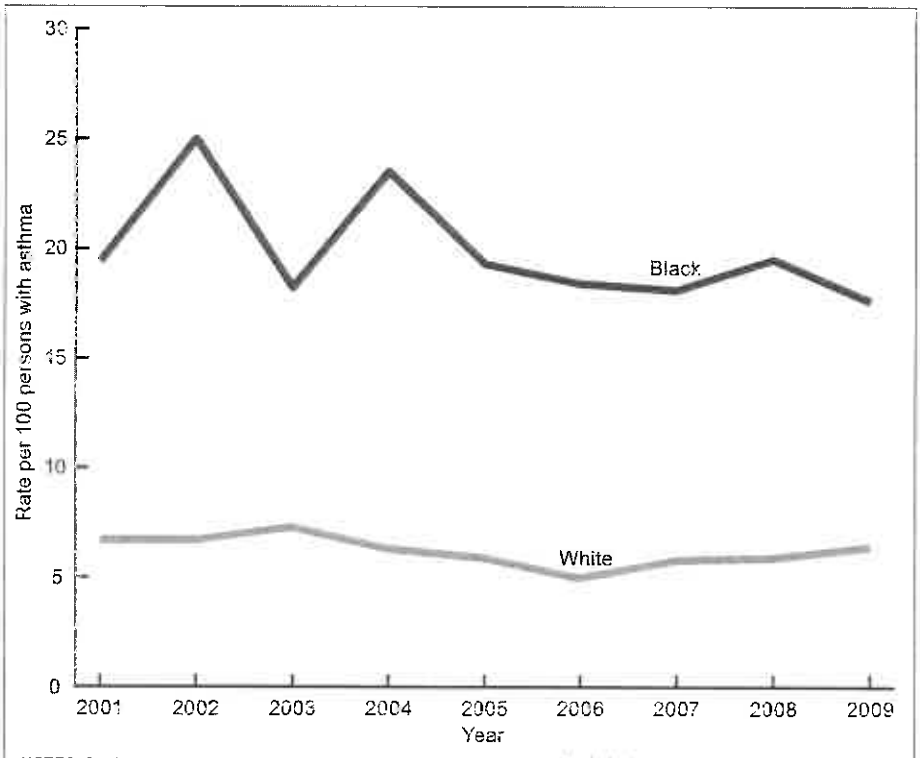
Differences in risk-based rates of ED visits, by race—The ED visit rate per 100 persons with asthma for black persons was more than twice as high as the rate for white persons during each year from 2001 to 2009 (Figure 16, Table 14).

Trends in risk-based rates of ED visits, by race—There was fluctuation but no significant trend in the ED visit rate per 100 persons with asthma for white and black persons from 2001 through 2009 (Figure 16, Table 14).



1 95% confidence interval.
NOTES: Crude (unadjusted) risk-based rates are presented. See Table 15 for underlying data.
SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

Figure 15. Asthma emergency department visit rates (risk-based), by age group, sex, race and ethnicity, and geographic region: United States, average annual 2007–2009

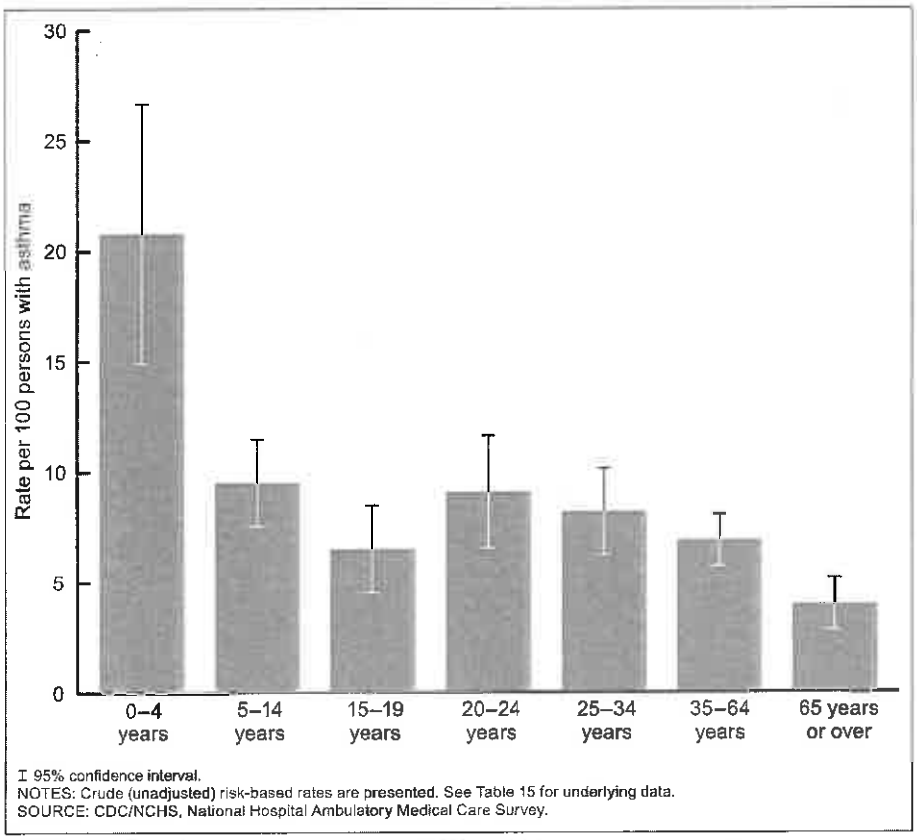


NOTES: Crude (unadjusted) risk-based rates are presented. See Table 14 for underlying data.
SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

Figure 16. Asthma emergency department visit rates (risk-based), by race: United States, 2001–2009

ED Visits by Age Group

Differences in risk-based rates of ED vsits, by detailed age group—The asthma ED visit rate was the highest for children aged 0–4 years (20.8 visits per 100 persons with asthma) and lowest for adults aged 65 or over (4.0 visits per 100 persons with asthma) (Figure 17, Table 15).



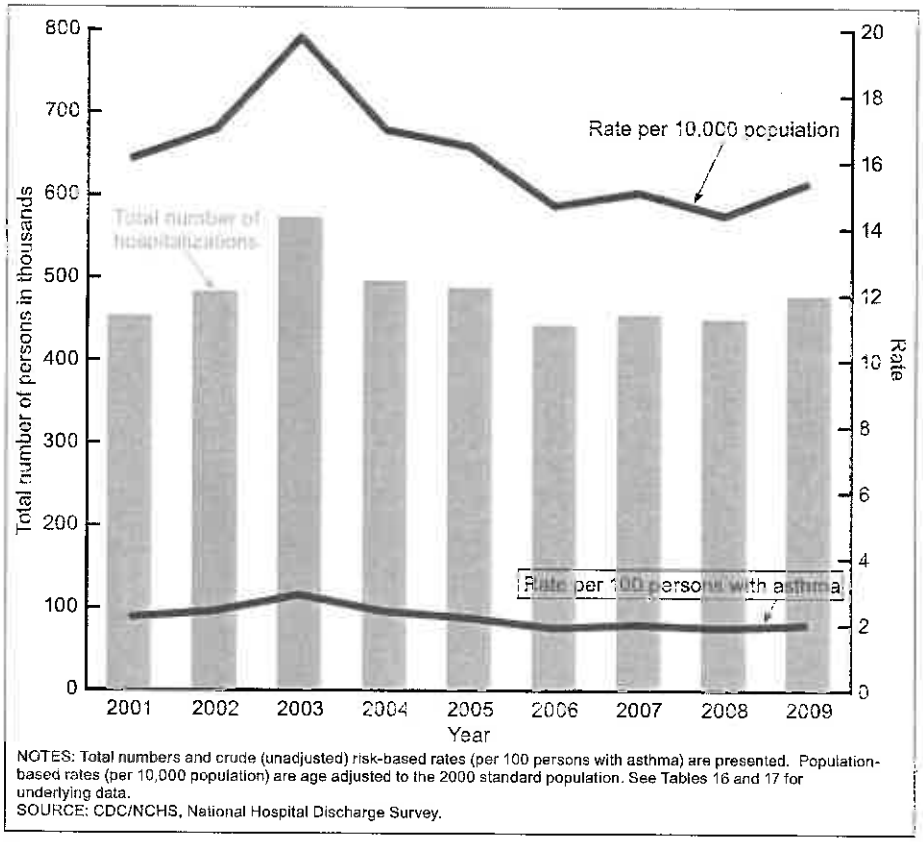
I 95% confidence interval.
NOTES: Crude (unadjusted) risk-based rates are presented. See Table 15 for underlying data.
SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

Figure 17. Asthma emergency department visit rates (risk-based), by detailed age group: United States, average annual 2007–2009

Hospital Inpatient Stays

Asthma hospitalization measures include the numbers and rates (population-based rates and risk-based rates) of discharges from nonfederal short-stay hospitals. Asthma hospitalizations represent a serious adverse outcome that is theoretically preventable with high-quality health care, patient education, and optimal management of asthma. Furthermore, an asthma hospitalization is a marker for increased risk of future asthma exacerbations (1).

This section presents 2001–2009 trends for asthma hospital inpatient stays for population-based rates, risk-based rates, and counts of hospitalizations. Differences between population subgroups are assessed over this period and for additional demographic detail using average annual estimates for the period 2007–2009. Additional detail not included in the figures can be found in Tables 16–18.



NOTES: Total numbers and crude (unadjusted) risk-based rates (per 100 persons with asthma) are presented. Population-based rates (per 10,000 population) are age adjusted to the 2000 standard population. See Tables 16 and 17 for underlying data.
SOURCE: CDC/NCHS, National Hospital Discharge Survey.

Figure 18. Asthma hospitalizations: United States, 2001–2009

Hospitalization Trends

Number of asthma hospitalizations—There was no significant trend in the total number of asthma hospitalizations from 2001 through 2009. In 2009, there were 479,300 hospitalizations for asthma (Figure 18).

Population-based rate of asthma hospitalizations—From 2001 through 2009, the trend in hospitalization rates was not significant (Figure 18, Table 16).

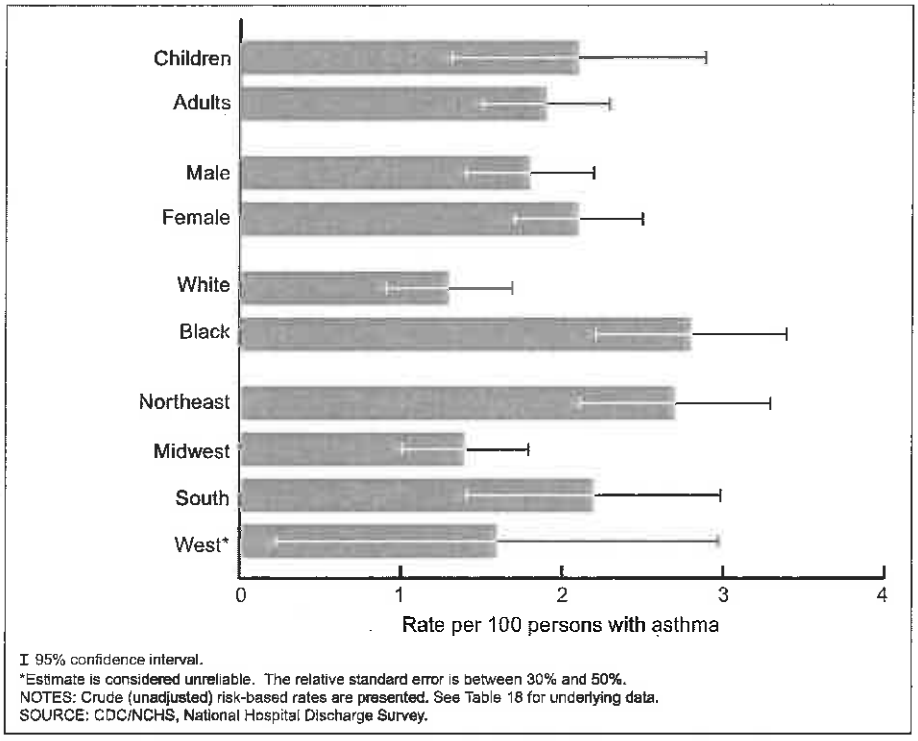
Risk-based rate of asthma hospitalizations—Hospitalization rates for persons with asthma did not follow a significant trend from 2001 through 2009 (Figure 18, Table 17).

Hospitalizations by Selected Characteristics

Differences in risk-based rates of asthma hospitalizations, by age group and sex—For the period 2007–2009, average annual asthma hospitalizations rates were similar between children aged 0–17 years and adults aged 18 and over (2.1 and 1.9 per 100 persons with asthma), and were similar between males and females (1.8 and 2.1 per 100 persons with asthma) (Figure 19, Table 18).

Differences in risk-based rates of asthma hospitalizations, by race—Black persons had an asthma hospitalization rate more than twice as high as white persons (2.8 compared with 1.3 hospitalizations per 100 persons with asthma) (Figure 19, Table 18).

Differences in risk-based rates of asthma hospitalizations, by geographic region—The rate of hospitalizations in the Northeast (2.7 per 100 persons with asthma) was higher than the rate in the Midwest (1.4 per 100 persons with asthma). There were no other statistically significant differences in hospitalization risk-based rates by geographic region (Figure 19, Table 18).



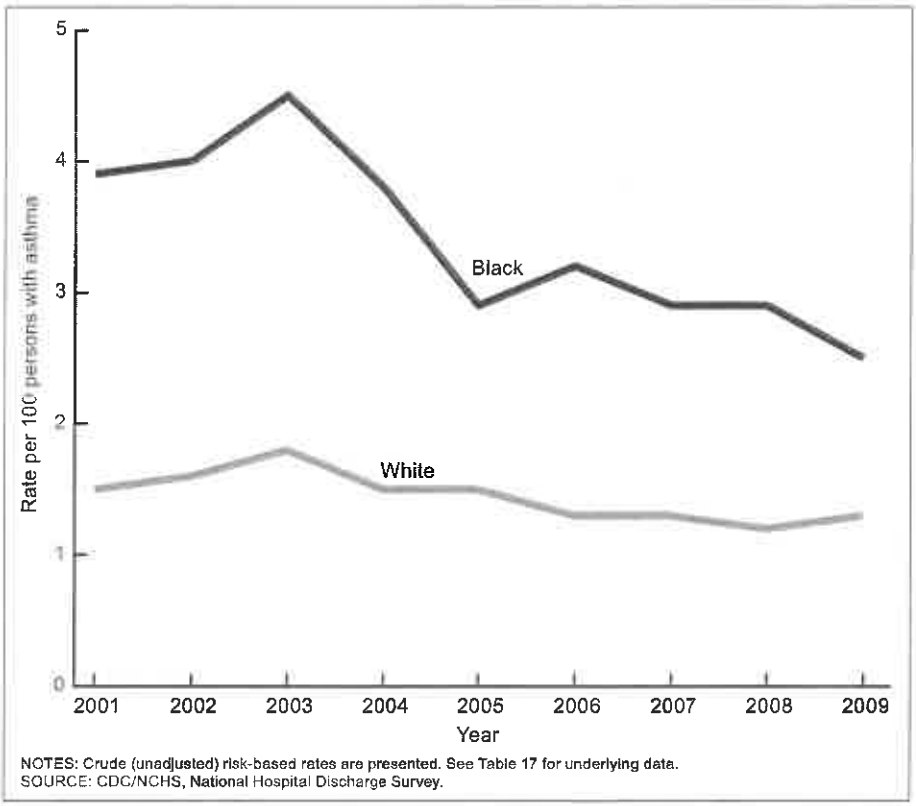
I 95% confidence interval.
*Estimate is considered unreliable. The relative standard error is between 30% and 50%.
NOTES: Crude (unadjusted) risk-based rates are presented. See Table 18 for underlying data.
SOURCE: CDC/NCHS, National Hospital Discharge Survey.

Figure 19. Asthma hospitalization rates (risk-based), by age group, sex, race, and geographic region: United States, average annual 2007–2009

Hospitalizations by Race

Differences in annual risk-based rates of asthma hospitalization, by race—The hospitalization rate per 100 persons with asthma among black persons was between 1.9 and 2.5 times higher than the rate for white persons during each year from 2001 to 2009 (Figure 20, Table 17).

Trends in risk-based rates of asthma hospitalization, by race—For black persons, the asthma hospitalization rate per 100 persons with asthma declined by an average rate of 6.3% per year from 2001 to 2009. For white persons, the asthma hospitalization rate declined by an average rate of 3.7% per year from 2001 to 2009 (Figure 20, Table 17).

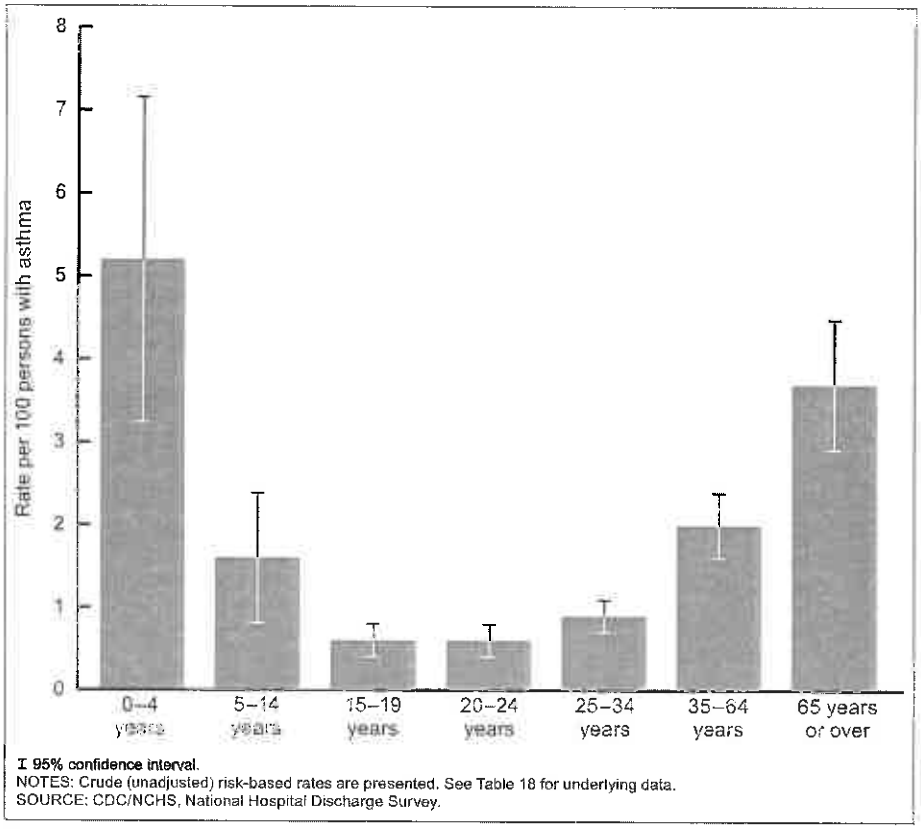


NOTES: Crude (unadjusted) risk-based rates are presented. See Table 17 for underlying data.
SOURCE: CDC/NCHS, National Hospital Discharge Survey.

Figure 20. Asthma hospitalization rates (risk-based), by race: United States, 2001–2009

Hospitalizations by Age

Differences in risk-based rates of asthma hospitalizations, by detailed age group—Children aged 0–4 years and adults aged 65 or over had higher asthma hospitalization rates (5.2 and 3.7 per 100 persons with asthma) than all other age groups. Persons aged 15–19 and 20–24 had similar rates, and both these age groups had lower rates than all older age groups. (Figure 21, Table 18).



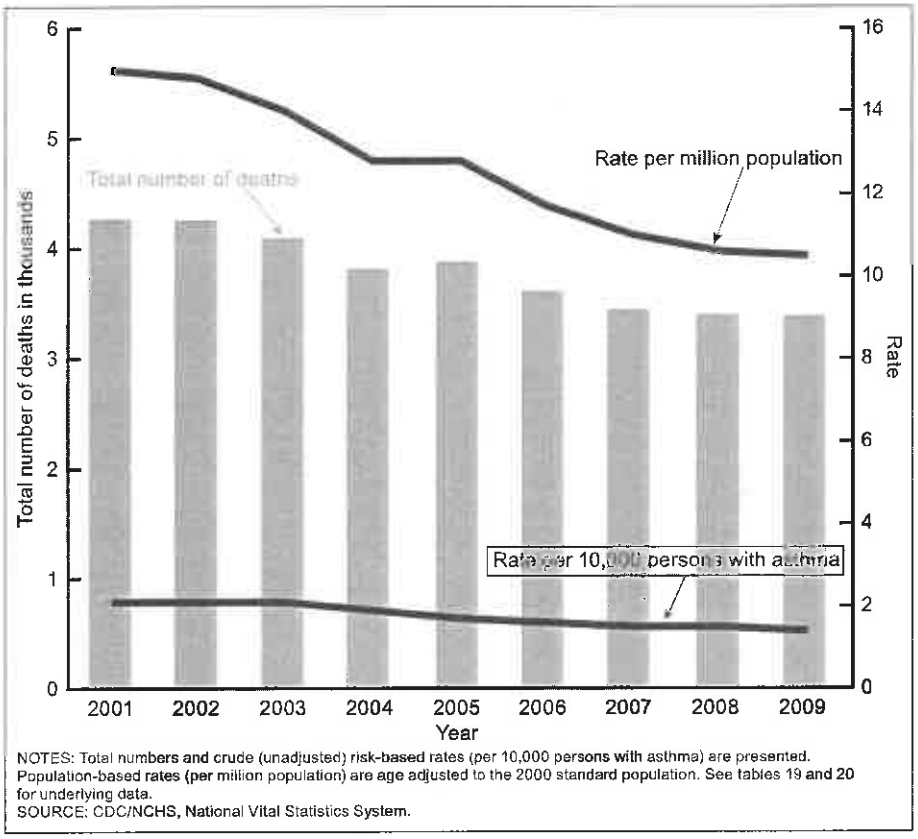
I 95% confidence interval.
NOTES: Crude (unadjusted) risk-based rates are presented. See Table 18 for underlying data.
SOURCE: CDC/NCHS, National Hospital Discharge Survey.

Figure 21. Asthma hospitalization rates (risk-based), by detailed age group: United States, average annual 2007–2009

Deaths

The number of asthma deaths and death rates (population-based rates and risk-based rates) are compiled from the complete set of death certificates filed in the 50 states and the District of Columbia. Asthma deaths are uncommon, especially among children and young adults, but they remain a focus of preventive efforts because high-quality health care and patient education should theoretically prevent asthma-related deaths (1). National asthma guidelines recommend early treatment and special attention to patients who are at high risk of asthma-related death (1). Predictors of death due to asthma include three or more ED visits for asthma in the past year, an asthma hospitalization or ED visit in the past month, overuse of short-acting beta agonist (short-term relief medication), a history of intubation or stay in an intensive care unit for asthma, difficulty perceiving asthma symptoms, lack of a written asthma action plan, certain patient characteristics (low socioeconomic status, female, nonwhite, current smoker, or major psychosocial problems), and the presence of other medical conditions such as cardiovascular disease (1).

This section presents 2001–2009 trends for asthma deaths for population-based rates, risk-based rates, and counts of deaths. Differences between population subgroups are assessed over this period and for additional demographic groups using the average annual estimates for the period 2007–2009. Additional detail not shown in the figures can be found in Tables 19–21.



NOTES: Total numbers and crude (unadjusted) risk-based rates (per 10,000 persons with asthma) are presented. Population-based rates (per million population) are age adjusted to the 2000 standard population. See tables 19 and 20 for underlying data.
SOURCE: CDC/NCHS, National Vital Statistics System.

Figure 22. Asthma deaths: United States, 2001–2009

Trends in Deaths

Number of asthma deaths—The number of asthma deaths declined steadily from 2001 (4,269) to 2009 (3,388) at a rate of 3.3% per year (Figure 22).

Population-based rate of deaths—Asthma deaths per million population declined 4.9% per year from 15.0 deaths per million population in 2001 to 10.5 deaths per million population in 2009 (Figure 22, Table 19).

Risk-based rate of deaths—Asthma deaths per 10,000 persons with asthma declined at a rate of 5.6% per year from 2.1 per 10,000 persons with asthma in 2001 to 1.4 per 10,000 persons with asthma in 2009 (Figure 22, Table 20).

Deaths by Selected Characteristics

Differences in risk-based rates of asthma deaths, by age group—Adults aged 18 and over had a higher asthma death rate (1.9 per 10,000 persons with asthma) compared with children aged 0–17 years (0.3 per 10,000 persons with asthma) (Figure 23, Table 21).

Differences in risk-based rates of asthma deaths, by sex—The death rate for females (1.6 per 10,000 persons with asthma) was higher compared with males (1.2 per 10,000 persons with asthma). This difference is driven primarily by the higher death rate for women than men among adults aged 65 and over (6.6 compared with 4.4 per 10,000 persons with asthma). For younger age groups, death rates for males were similar to or higher than rates for females (Figure 23, Table 21).

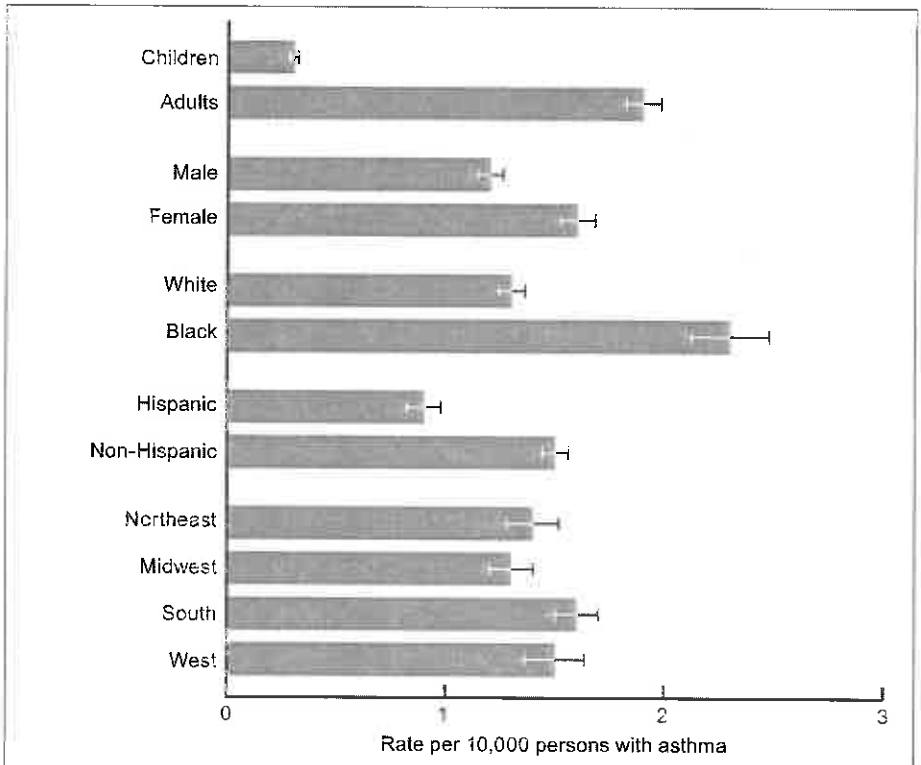
Differences in risk-based rates of asthma deaths, by race and ethnicity—Black persons had a higher asthma death rate compared with white persons (2.3 compared with 1.3 per 10,000 persons with asthma). Non-Hispanic persons had a higher death rate compared with Hispanic persons (1.5 compared with 0.9 per 10,000 persons with asthma) (Figure 23, Table 21).

Differences in risk-based rates of asthma deaths, by geographic region—The asthma death rates in the West and the South (1.5 and 1.6 per 10,000 persons with asthma) were higher than the asthma death rate in the Midwest region (1.3 per 10,000 persons with asthma) (Figure 23, Table 21).

Deaths by Race and Ethnicity

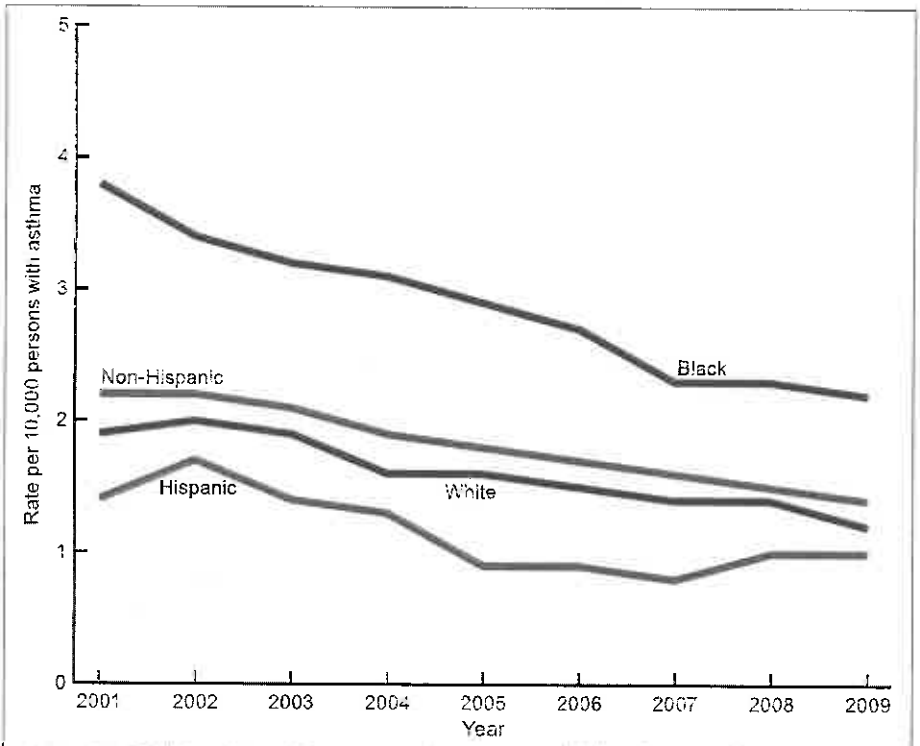
Differences in annual risk-based rates of asthma deaths, by race and ethnicity

The death rate per 10,000 persons with asthma for black persons was 1.6 to 2.0 times higher than the rate for white persons during each year from 2001 to 2009. The asthma death rate for non-Hispanic persons was 1.3 to 2.0 times higher than the rate for Hispanic persons during each year (Figure 24, Table 20).



I 95% confidence interval.
NOTES: Crude (unadjusted) risk-based rates are presented. See Table 21 for underlying data.
SOURCE: CDC/NCHS, National Vital Statistics System.

Figure 23. Asthma death rates (risk-based), by age group, sex, race and ethnicity, and geographic region: United States, average annual 2007–2009



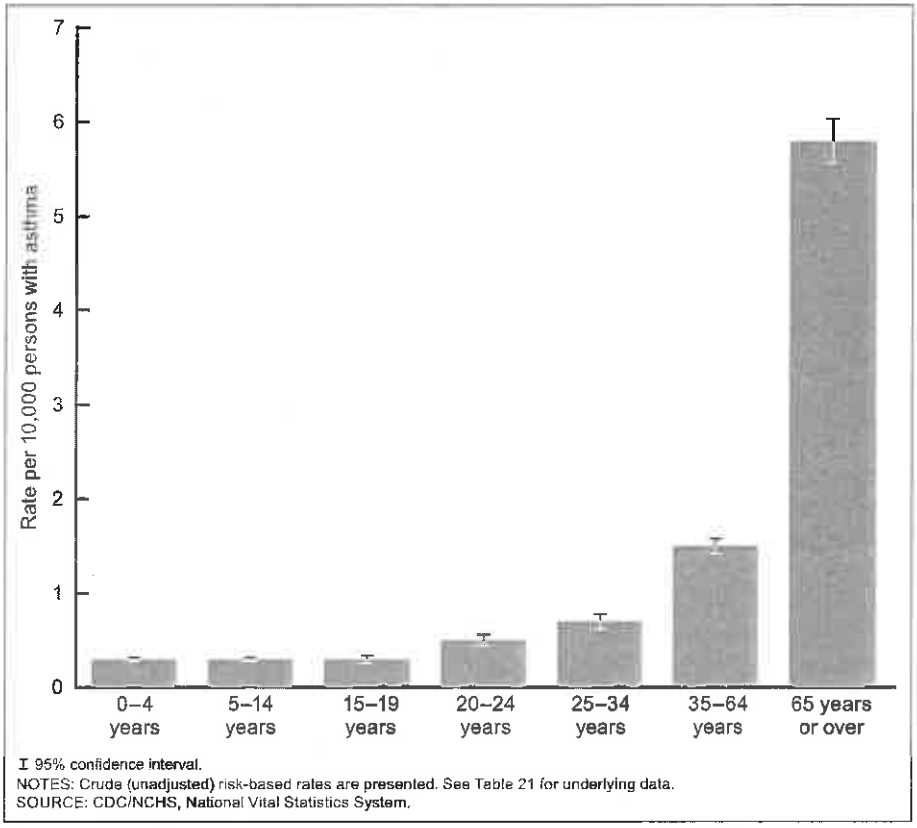
NOTES: Crude (unadjusted) risk-based rates are presented. See Table 20 for underlying data. Categories are not mutually exclusive. Persons are identified as either black or white and either as Hispanic or non-Hispanic.
SOURCE: CDC/NCHS, National Vital Statistics System.

Figure 24. Asthma death rates (risk-based), by race and ethnicity: United States, 2001–2009

Trends in risk-based rates of asthma deaths, by race and ethnicity—From 2001 to 2009, asthma death rates per 10,000 persons with asthma declined at a rate of 6.7% per year for black persons, and at a rate of 5.6% per year for white persons. During this same period, death rates declined at a rate of 5.8% for non-Hispanic persons and 7.1% per year for Hispanic persons (Figure 24, Table 20).

Deaths by Age

Differences in risk-based rates of asthma deaths, by detailed age group—For the period 2007–2009, the average annual asthma death rates for the three youngest age groups (0–4, 5–14, and 15–19 years) were the same (0.3 per 10,000 persons with asthma). The asthma death rate for adults aged 65 or over was the highest at 5.8 per 10,000 persons with asthma. The rate for adults aged 35–64 (1.5 per 10,000 persons with asthma) was higher compared with all younger age groups, as were the rates for adults aged 25–34 (0.7 per 10,000 persons with asthma) and adults aged 20–24 (0.5 per 10,000 persons with asthma) (Figure 25, Table 21).



I 95% confidence interval.
NOTES: Crude (unadjusted) risk-based rates are presented. See Table 21 for underlying data.
SOURCE: CDC/NCHS, National Vital Statistics System.

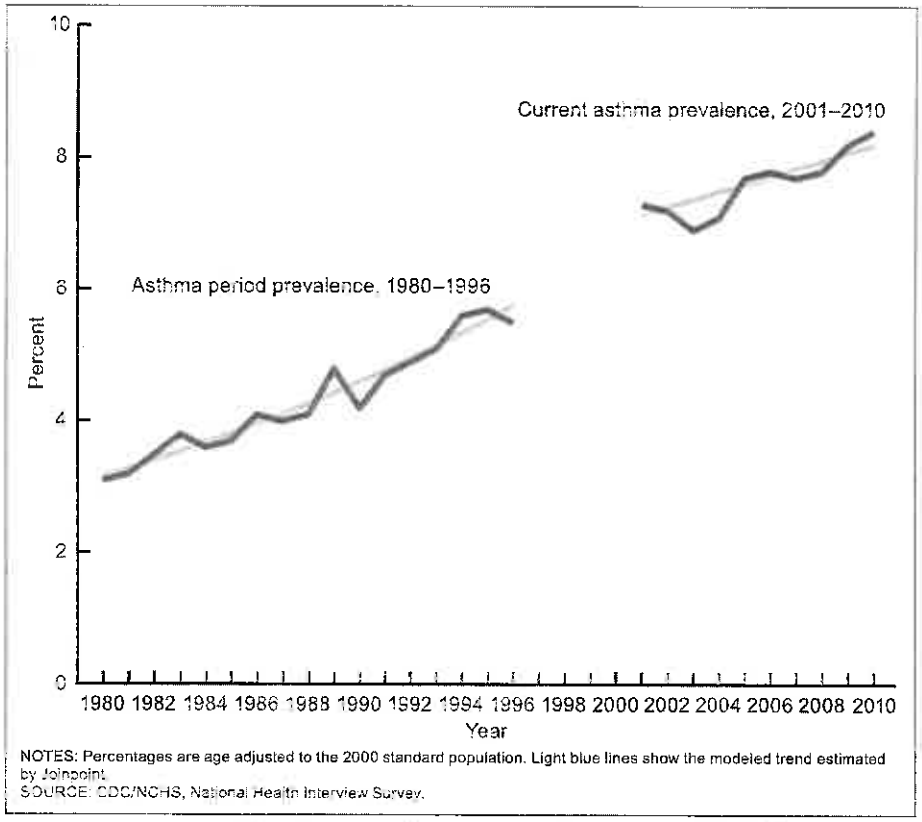
Figure 25. Asthma death rates (risk-based), by detailed age group: United States, average annual 2007–2009

Asthma Trends in the United States From 1980

Over the past two decades, advances in research, medications, treatment approach, public health interventions, and public awareness have all affected how asthma is diagnosed, managed, and perceived. This section presents trends from 1980 in asthma prevalence, hospitalizations, and deaths to provide greater context for the trends (since 2001) presented in the preceding sections. Disparities by sex and race are also shown for long-term trends for asthma hospitalizations and deaths. The most recent data for morbidity measures and asthma management strategies used to track Healthy People 2020 objectives are also discussed.

Historical Perspective

The long-term trend in asthma prevalence from 1980 to 2010 is shown in Figure 26. Asthma period prevalence was the original measure of U.S. asthma prevalence and estimated the percentage of the population that had asthma in the previous 12 months. Estimates were based on the question, "In the past 12 months, did anyone in the family have asthma?" Estimates rose from 3.1% in 1980 to 5.5% in 1996. In 1997, a major redesign of NHIS resulted in a break in the trend data due to a change in the questions used to estimate asthma prevalence (14). Beginning in 1997, two questions were used to measure the percentage of the population affected by asthma. The first question asked if a physician had ever diagnosed asthma. The second question asked about the occurrence of an asthma attack in the past 12 months (asthma attack prevalence). From 1997 to 2000, the measure of asthma attack prevalence was the available estimate of recent, active asthma but it excluded persons with well-controlled asthma. In 2001, current asthma prevalence (measured by the question, "Do you still have asthma?" for those with an asthma diagnosis) was introduced to include all



NOTES: Percentages are age adjusted to the 2000 standard population. Light blue lines show the modeled trend estimated by Joinpoint.
SOURCE: CDC/NCHS, National Health Interview Survey.

Figure 26. Asthma period prevalence and current asthma prevalence: United States, 1980–2010

persons with asthma. However, current asthma prevalence estimates from 2001 onward are point prevalence estimates and therefore are not directly comparable with asthma period prevalence estimates from 1980 to 1996 (see table in Appendix III for NHIS question wording for each type of prevalence estimate). Current asthma prevalence estimates are considered more valid than asthma period prevalence because they rely on receiving a diagnosis of asthma from a health professional. Current asthma prevalence increased from 7.3% in 2001 to 8.4% in 2010, an increase of 1.5% per year. This rate of increase was lower than the rate of increase for asthma period prevalence from 1980 to 1996 (3.8% per year).

Population-based outcome rates show the overall trend and impact on the general population. Risk-based rates are informative because they control for prevalence differences between groups. However, analyses of risk-based rates over longer periods are complicated by the change in asthma prevalence measurement over time, as discussed above. These changes preclude direct comparison of risk-based rates over the period 1980–2009. Therefore, population-based rates are presented in the following discussion.

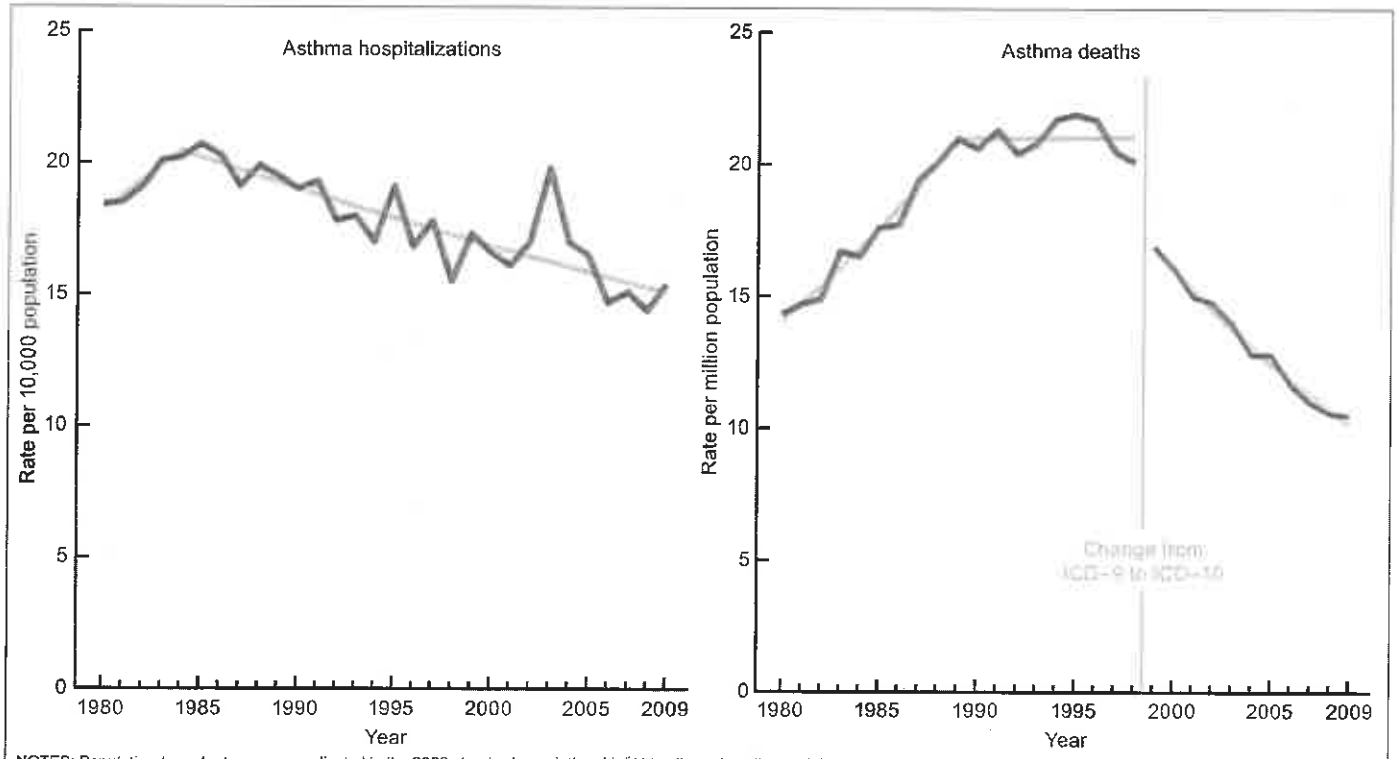
During the rise of asthma prevalence in the 1980s and 1990s, adverse outcomes were also increasing. The increase in asthma mortality in the 1980s led to a major public health response, including the formulation and dissemination of asthma treatment guidelines, asthma surveillance, and coordinated interventions on local and state levels (1,7,15,16). Figure 27 shows the long-term trends in population-based rates of asthma hospitalizations and deaths. Asthma hospitalization rates increased at an average rate of 2.9% per year from 1980 to 1984 and then decreased by an average rate of 1.2% per year from 1984 to 2009. Asthma death rates increased by an average rate of 4.6% per year from 1980 to 1989 and then remained stable from 1989 to 1998. Asthma death rates decreased by an average rate of 4.9% per year from 1999 to 2009. Death rates from before and after 1999 are not directly comparable due to the transition from *International Classification of Diseases, Ninth Revision* (ICD–9) to ICD–10 in 1999 (see Appendix III: Definition of Terms for details). Analysis of the impact of the transition to ICD–10 indicated that an estimated 11% of the decline in deaths between 1998 and 1999 could be attributed to the coding change alone. The continuing decline after 1999 should not have been influenced by the change in the classification system.

Analyses of trends since 2001 in earlier sections of this report show that both population-based rates and risk-based rates for asthma hospitalizations, hospital outpatient department visits, and ED visits did not change significantly over this recent period, whereas both population-based rates and risk-based rates for asthma hospitalizations and deaths declined. These trends indicate that on a national level, earlier progress in reducing the impact of these serious asthma outcomes has continued. Reducing the less severe, but vastly more common morbidity associated with asthma remains a challenge. The Guidelines for the Diagnosis and Management of Asthma recommend routine visits every 6 months, regardless of asthma severity, as a means to continually assess and improve asthma control (1).

Disparities in Asthma Hospitalization and Death Rates

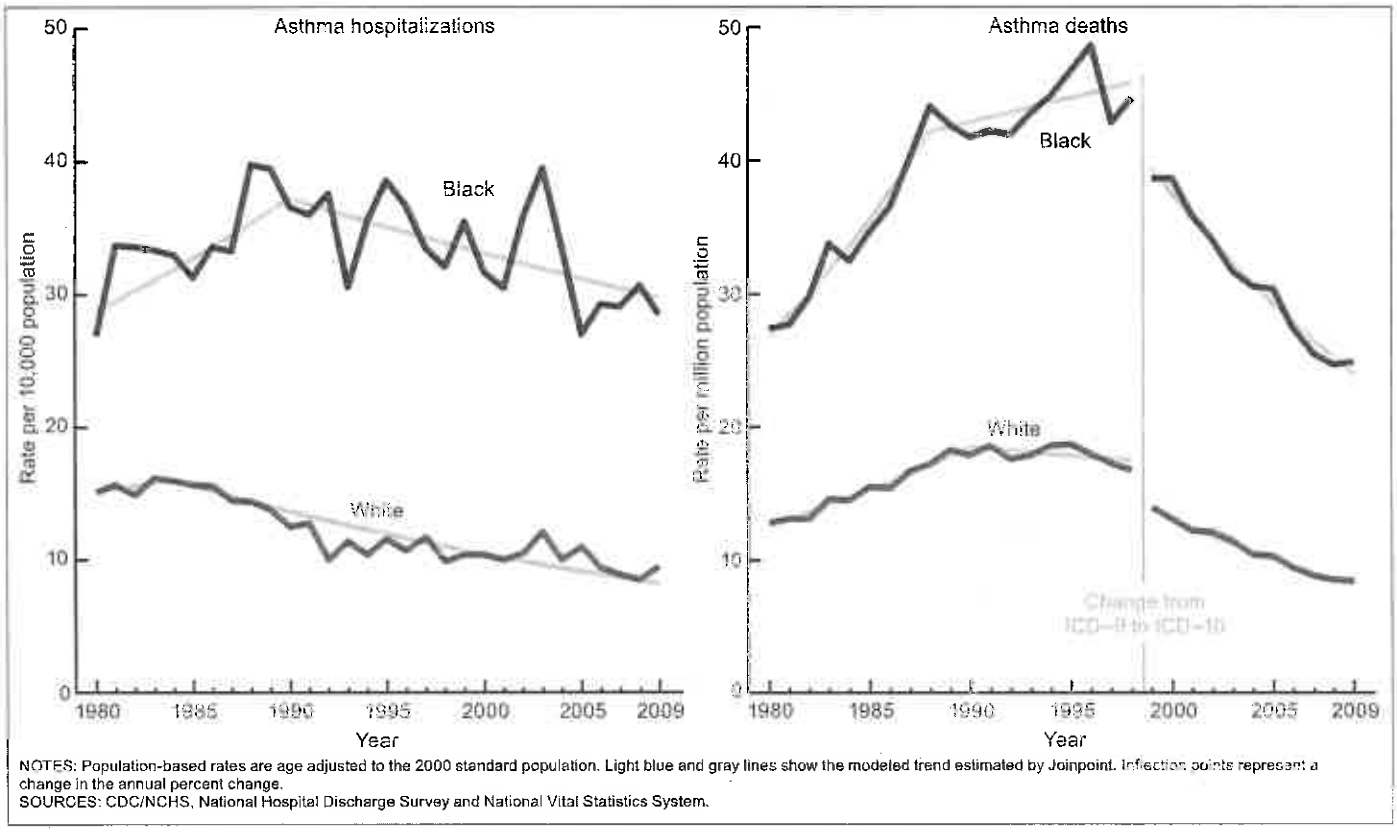
Racial disparities in asthma hospitalizations and deaths have been historically large (two to three times higher among black persons compared with white persons) and remain so despite progress in lowering these adverse outcomes for the overall population. Figure 28 shows trends by race in population-based hospitalization and death rates.

From 1980 to 1985, there was no significant trend in the asthma hospitalization rate for white persons. From 1985 to 2009, the rate for white persons decreased 2.7% per year. For black persons, the asthma hospitalization rate increased 2.6% per year from 1980 to 1990, and then decreased 1.2% per year from 1990 to 2009. At the end of the period in 2009, the rate ratio for



NOTES: Population-based rates are age adjusted to the 2000 standard population. Light blue lines show the modeled trend estimated by Joinpoint. Inflection points represent a change in the annual percent change.
SOURCES: CDC/NCHS, National Hospital Discharge Survey and National Vital Statistics System.

Figure 27. Asthma hospitalization rates and asthma death rates (population-based): United States, 1980–2009



NOTES: Population-based rates are age adjusted to the 2000 standard population. Light blue and gray lines show the modeled trend estimated by Joinpoint. Inflection points represent a change in the annual percent change.
SOURCES: CDC/NCHS, National Hospital Discharge Survey and National Vital Statistics System.

Figure 28. Asthma hospitalization rates and asthma death rates (population-based), by race: United States, 1980–2009

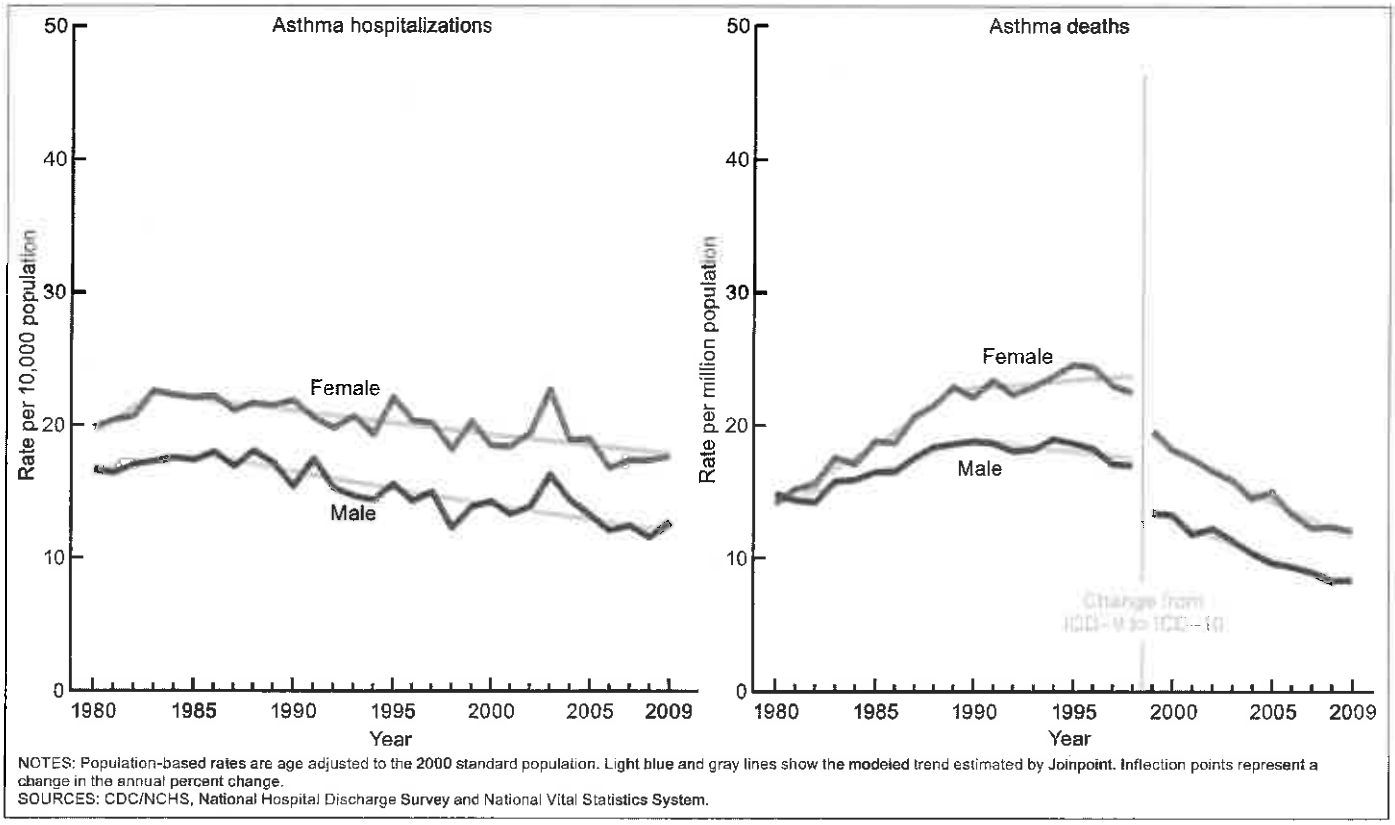
black to white hospitalization rates was 3.0 compared with a black to white rate ratio of 3.0 in 2000, 2.9 in 1990, and 1.8 in 1980. Although hospitalization rates decreased for both black and white persons after 1990, the relative difference in the rate for black persons compared with white persons was about the same in 2009 as it was in 2000 and in 1990. The absolute difference in 2009 was larger than it was in 1980 because of the increase in rates for black persons between 1980 and 1990.

Asthma death rates for black persons increased by an average rate of 5.8% per year from 1980 to 1988; the trend from 1988 to 1998 did not reach statistical significance. For white persons, asthma death rates increased by an average rate of 3.9% per year from 1980 to 1990, and remained stable from 1990 to 1998. After implementation of ICD–10 in 1999, asthma death rates for black persons decreased by an average rate of 4.9% per year, and decreased for white persons by 5.2% per year (note that although the absolute decline in the asthma death rate for black persons was greater than for white persons, the average percent change was smaller). The asthma death rate was three times higher for black persons compared with white persons in 2009.

These patterns show that much progress has been made in lowering asthma death rates overall. However, black persons continue to have a much higher risk of hospitalization and death from asthma compared with white persons.

Differences also existed in rates for asthma outcomes by sex. Females had higher rates of asthma hospitalizations and deaths compared with males (Figure 29). Asthma hospitalization rates for females increased by an average rate of 4.9% per year from 1980 to 1983, then decreased by an average rate of 0.9% per year from 1983 to 2009. For males, the asthma hospitalization rate increased by an average rate of 1.0% per year from 1980 to 1986, and then decreased by an average rate of 1.6% per year from 1986 to 2009. The hospitalization rate for females was 40% higher compared with the rate for males in 2009.

Prior to the 1999 implementation of ICD–10 for cause of death classification, asthma death rates increased by an average rate of 5.3% per year for females from 1980 to 1989, but there was no statistically significant change from 1989 to 1998. For males, asthma death rates increased by an average rate of 3.0% per year from 1980 to 1990, and then decreased by an average rate of 1.1% per year from 1990 to 1998. From 1999 to 2009, asthma death rates declined by an average rate of 4.9% per year for females, and by an average rate of 5.0% per year for males. In 2009, the ratio of female to male asthma death rates was 1.4. Thus, the relative disparity between females and males that developed in the 1980s remains, despite progress in reducing asthma death rates for both sexes over the past decade.

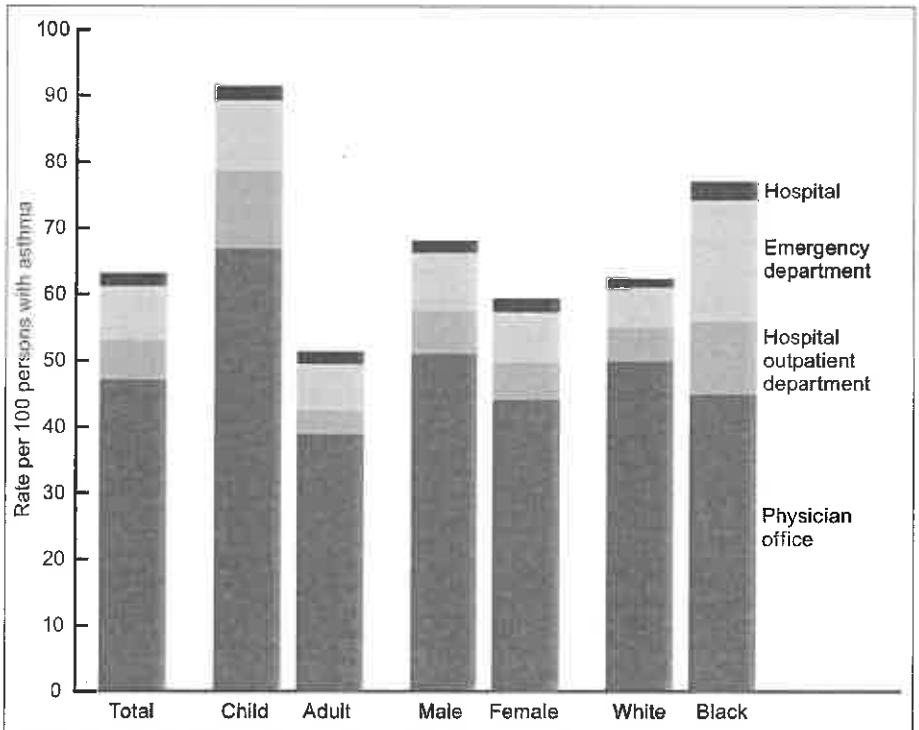


NOTES: Population-based rates are age adjusted to the 2000 standard population. Light blue and gray lines show the modeled trend estimated by Joinpoint. Inflection points represent a change in the annual percent change.
SOURCES: CDC/NCHS, National Hospital Discharge Survey and National Vital Statistics System.

Figure 29. Asthma hospitalization rates and asthma death rates (population-based), by sex: United States, 1980–2009

Although progress has been made in decreasing rates for asthma hospitalizations and asthma deaths, and in decreasing absolute race and sex disparities, relative disparities persist. Some, but not all, of the disparities by race and sex are explained by higher current asthma prevalence among black persons and females. For example, for asthma hospitalization rates in 2008, the risk-based rate ratio of black to white persons with asthma was 2.4, compared with the population-based rate ratio of 3.6. For asthma death rates, the 2008 risk-based rate ratio of black to white persons with asthma was 2.3, compared with the population-based rate ratio of 2.9. Yet, even after accounting for differences in asthma prevalence between groups by assessing rate ratios for risk-based rates for the most recently available data, the relative disparities persist.

Risk-based rates for each type of asthma health care encounter for the three main disparate groups (adult and child, male and female, and white and black persons) provide a perspective on the relative burden for each type of encounter (Figure 30). Hospitalizations represent



NOTES: Crude (unadjusted) rates are presented. See Tables 11, 12, 15, and 18 for underlying data.
SOURCES: CDC/NCHS, National Ambulatory Medical Care Survey, National Hospital Ambulatory Medical Care Survey, and National Hospital Discharge Survey.

Figure 30. Asthma physician office visit rates, hospital outpatient department visit rates, emergency department visit rates, and hospitalization rates (risk-based), by age, sex, and race: United States, average annual 2007–2009

Table A. School and work absences and percentage with activity limitation caused by asthma, among persons with at least one asthma attack in the past year: United States, 2008

Population	Population in millions (weighted estimate)	Total days missed[1] in millions (standard error)	Percentage with 1 or more asthma absence days in the past year[1] (standard error)	Percentage with activity limitation caused by asthma or breathing problem (standard error)
Children aged 5–17 years	3.3	10.5 (1.3)	58.7 (3.6)	5.5 (1.2)
Adults aged 18 or over, currently employed	4.9	14.2 (4.7)	33.5 (2.8)	6.2 (1.3)
Adults aged 18 or over, not currently employed	3.8	22.0 (5.9)	29.1 (3.1)	26.5 (2.7)

[1]Excludes records with missing responses for days of school (1.0% among children aged 5–17 years), work (0.4% among adults currently employed), or household or other activity missed due to asthma (25.1% among adults not currently employed).

SOURCE: CDC/NCHS National Health Interview Survey.

Table B. Percentage ever receiving asthma self-management education among persons with current asthma: percent (standard error): United States, 2003 and 2008

Management Strategy	2003	2008
Given an asthma action plan	33.6 (1.2)	34.2 (1.2)
Taken a class to learn how to manage asthma	12.4 (0.9)	12.2 (0.8)
Taught to recognize early signs and symptoms of an asthma attack	55.1 (1.3)	59.9 (1.3)[1]
Taught to respond to an asthma attack	64.8 (1.3)	68.2 (1.3)
Taught how to use a peak flow meter	45.1 (1.3)	42.2 (1.3)
Given advice on environmental control[2]	48.3 (1.3)	50.9 (1.4)

[1]Differs from 2003 value (chi square test p value < 0.05).

[2]Percentage reporting that a health professional gave them advice on how to change things in their home, school, or work environment to improve their asthma symptoms.

SOURCE: CDC/NCHS, National Health Interview Survey.

the most costly encounter and indicate a severe exacerbation, but are the least common type of encounter. ED visits also represent potentially severe exacerbations but could include visits for primary care as well. Visits to primary care settings (physician offices and hospital outpatient departments) represent the vast majority of health care use for persons with asthma. There was an average of 63.2 health care encounters per 100 persons with asthma for the period 2007–2009. Of these, 47.1 were visits to physician offices, 6.0 were visits to hospital outpatient departments, 8.1 were visits to EDs, and 2.0 were hospitalizations. Children aged 0–17 years had more health care visits than adults aged 18 and over (91.5 and 51.4 per 100, respectively). Except for asthma hospitalizations, children aged 0–17 years had significantly higher risk-based visit rates in every setting compared with adults aged 18 and over. Males and females had a similar number of health care visits (68.1 and 59.4 per 100, respectively), and there were no differences in rates of health care encounters by sex for any setting. Black and white persons had a similar overall number of health care visits (77.2 and 62.4 per 100, respectively), but black persons had higher ED visit and hospitalization rates per 100 persons with asthma compared with white persons. This pattern was also reported in the previous asthma surveillance report (2). Risk-based rates for visits in primary care settings overall (physician offices and hospital outpatient departments) were similar by race, but black persons had higher visit rates to hospital outpatient departments and lower visit rates to private physician offices compared with white persons. The recent pattern of similar primary care visit rates between races differs from patterns observed in earlier periods with higher primary care visit rates among white persons compared with black persons with asthma (2).

Other Measures of Asthma Morbidity

In addition to health care utilization rates, estimates of school and work absences and activity limitations caused by asthma provide measures of the impact of asthma in the United States. Table A shows school and work absences among persons with at least one asthma attack in the previous 12 months for 2008. There were 10.5 million school days missed due to asthma among children aged 5–17 years, 14.2 million work days missed among currently employed adults aged 18 and over, and 22.0 million missed days of housework and other activity among adults not currently employed.

Recent Trends in Use of Asthma Management Tools

An important strategy to prevent asthma exacerbations and morbidity is asthma education. The percentage of persons with current asthma who reported receipt of asthma self-management education and tools for 2003 and 2008 is shown in Table B. These data are from periodic NHIS asthma supplements, which are used to measure progress meeting Healthy People respiratory health objectives. The 2008 supplement is the most recent. There was little change in these proportions between 2003 and 2008. The National Asthma Education and Prevention Program (NAEPP) guidelines recommend that all asthma patients have written instructions for daily management of asthma, for how to recognize and respond to worsening asthma symptoms, and for medication dosage and frequency of its administration, but only 34% of persons with current asthma reported receiving an asthma action plan (1). About 12% of persons with current asthma had taken a

class to learn how to manage asthma. Higher percentages were taught how to recognize (55%–60%) and respond (65%–68%) to signs and symptoms of an asthma attack, and these indicators of asthma education increased between 2003 and 2008. A lower percentage reported being taught how to use a peak flow meter (42%–45%) and receiving advice on environmental control at home, school, or work (48%–51%).

Data about use of evidence-based asthma management strategies were not collected in NHIS before the release of the 1991 NAEPP guidelines, which focused treatment on the central role of inflammation in asthma morbidity. Therefore, longer-term changes in treatment as reported by people with asthma cannot be directly analyzed with NHIS data.

Federal Efforts to Address Asthma

Asthma has been a focus for public health action after increasing asthma prevalence and mortality rates were recognized in the 1980s. Healthy People goals for asthma have been expanded over three decades. Asthma-specific Healthy People 2020 Objectives include: 1) reducing asthma deaths, 2) reducing hospitalizations for asthma, 3) reducing ED visits for asthma, 4) reducing activity limitations among persons with current asthma, 5) reducing the proportion of persons with asthma who miss school or work days, 6) increasing the proportion of persons with current asthma who receive formal patient education, 7) increasing the proportion of persons with current asthma who receive appropriate asthma care according to NAEPP, and 8) increasing the number of states and territories (including the District of Columbia) with a comprehensive surveillance system for tracking asthma cases, illness, and disability at the state level. Federal agencies have developed a collaborative approach to address these objectives.

NAEPP was established in 1989 by the National Heart, Lung and Blood Institute (NHLBI) as a collaborative effort between major scientific, professional, governmental, and voluntary organizations. NAEPP was designed to increase awareness of asthma as a significant public health problem, to educate the public and health professionals about asthma, to develop clinical treatment guidelines, and to promote research. NAEPP guidelines have been periodically updated since their first publication in 1991, and most recently in 2007.

CDC's National Asthma Control Program (NACP) was created in 1999 to align public health initiatives with Healthy People objectives. NACP has three major activities: conducting surveillance, promoting evidence-based interventions, and building partnerships.

Surveillance activities include building systems to provide detailed and local data on asthma prevalence, impact, and management such as including asthma modules in the Behavioral Risk Factor Surveillance System (17); developing the National Asthma survey (http://www.cdc.gov/asthma/ACBS.htm) (18); and funding (with the NHLBI) the periodic inclusion of asthma questions in NHIS to monitor progress toward Healthy People goals (19). NACP works to establish and maintain a network of asthma public health partners, including state health departments, local entities (schools, city health departments, and hospitals), and other federal agencies to achieve asthma control objectives (6).

Conclusion

Over the past few decades, the prevalence of asthma in the United States has grown. Since 2001, the number of persons with asthma has continued to increase. There have not, however, been comparable increases in the number of health care encounters for asthma. Since rising asthma mortality was first recognized in the 1980s, the number and rate for this most severe adverse asthma outcome have declined dramatically. Population-based rates for asthma hospitalizations have declined since the mid-1980s and rates for deaths have declined since 1999.

However, for many outcomes, racial and gender disparities remain, and in some cases have increased. In addition to disparities in outcomes, there are other areas of asthma care that still require concerted public health attention. The prevalence of asthma attacks among persons with asthma, although declining, remains above 50%, with only one in eight persons reporting taking a formal asthma management education course and only one in three reporting having asthma management plans. There are an estimated 46.7 million lost school, work, and activity days per year. The more recent plateau in the uptake of effective asthma-management strategies highlights remaining challenges in addressing the burden of asthma in the United States.

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Table 1. Percentage with current asthma, by year and selected characteristics: United States, 2001–2010

Characteristic	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
	Percent (standard error)[1]									
Total	7.3 (0.1)	7.2 (0.2)	6.9 (0.2)	7.1 (0.2)	7.7 (0.2)	7.8 (0.2)	7.7 (0.2)	7.8 (0.2)	8.2 (0.2)	8.4 (0.2)
Sex										
Male	6.3 (0.2)	6.1 (0.2)	5.8 (0.2)	6.3 (0.2)	6.4 (0.2)	7.0 (0.3)	6.5 (0.2)	7.1 (0.3)	7.1 (0.3)	7.0 (0.2)
Female	8.3 (0.2)	8.1 (0.2)	7.9 (0.2)	7.8 (0.2)	8.8 (0.2)	8.6 (0.3)	8.8 (0.3)	8.5 (0.3)	9.2 (0.3)	9.8 (0.3)
Race[2]										
White	7.2 (0.2)	6.9 (0.2)	6.6 (0.2)	6.9 (0.2)	7.4 (0.2)	7.6 (0.2)	7.4 (0.2)	7.4 (0.2)	7.8 (0.2)	7.8 (0.2)
Black	8.4 (0.4)	9.2 (0.4)	8.8 (0.4)	8.9 (0.5)	9.5 (0.4)	9.0 (0.4)	9.8 (0.5)	10.0 (0.5)	10.8 (0.6)	11.9 (0.5)
Other races	7.2 (0.5)	6.3 (0.5)	6.4 (0.6)	6.3 (0.5)	7.1 (0.6)	8.3 (0.7)	6.9 (0.6)	7.9 (0.7)	7.6 (0.7)	8.1 (0.6)
Ethnicity[3]										
Hispanic or Latino	5.8 (0.3)	4.7 (0.3)	5.4 (0.3)	5.2 (0.3)	6.2 (0.3)	6.2 (0.4)	6.9 (0.4)	5.9 (0.4)	6.4 (0.4)	7.2 (0.4)
Not Hispanic or Latino	7.6 (0.2)	7.5 (0.2)	7.2 (0.2)	7.5 (0.2)	7.9 (0.2)	8.1 (0.2)	7.9 (0.2)	8.2 (0.2)	8.6 (0.2)	8.7 (0.2)
Age										
Children (0–17)	8.7 (0.3)	8.3 (0.3)	8.5 (0.3)	8.5 (0.3)	8.9 (0.3)	9.3 (0.4)	9.1 (0.4)	9.4 (0.4)	9.6 (0.4)	9.3 (0.3)
Adults (18 or over)	6.9 (0.2)	6.8 (0.2)	6.4 (0.2)	6.7 (0.2)	7.2 (0.2)	7.3 (0.2)	7.3 (0.2)	7.3 (0.2)	7.7 (0.2)	8.2 (0.2)
Age group										
0–4 years	5.7 (0.4)	6.0 (0.5)	5.9 (0.5)	5.6 (0.4)	6.8 (0.5)	5.8 (0.6)	6.8 (0.6)	6.2 (0.6)	6.3 (0.6)	6.0 (0.6)
5–14 years	9.9 (0.4)	9.2 (0.4)	9.5 (0.4)	9.1 (0.4)	9.8 (0.4)	10.9 (0.6)	9.4 (0.5)	10.9 (0.6)	11.2 (0.5)	10.7 (0.5)
15–34 years	8.0 (0.3)	7.6 (0.3)	7.2 (0.3)	7.1 (0.3)	7.5 (0.3)	7.8 (0.4)	8.4 (0.4)	8.1 (0.4)	7.8 (0.4)	8.6 (0.3)
35–64 years	6.7 (0.2)	6.8 (0.2)	6.4 (0.2)	6.8 (0.2)	7.1 (0.2)	7.3 (0.3)	6.9 (0.3)	7.1 (0.3)	7.9 (0.3)	8.1 (0.3)
65 years or over	6.0 (0.4)	5.9 (0.4)	5.8 (0.3)	7.0 (0.4)	7.6 (0.4)	7.0 (0.5)	7.5 (0.5)	7.2 (0.4)	7.7 (0.5)	8.1 (0.4)
Region										
Northeast	8.3 (0.4)	8.2 (0.4)	7.7 (0.4)	7.4 (0.4)	8.7 (0.4)	9.1 (0.5)	8.5 (0.5)	8.2 (0.5)	9.4 (0.5)	8.8 (0.5)
Midwest	7.5 (0.3)	7.3 (0.3)	7.6 (0.3)	7.6 (0.3)	7.7 (0.3)	8.3 (0.5)	8.2 (0.5)	8.6 (0.4)	8.9 (0.5)	9.6 (0.4)
South	7.1 (0.2)	6.7 (0.2)	6.3 (0.2)	6.7 (0.2)	7.2 (0.2)	7.3 (0.3)	7.3 (0.3)	7.0 (0.3)	7.5 (0.3)	8.3 (0.3)
West	6.7 (0.3)	7.1 (0.3)	6.5 (0.3)	7.3 (0.3)	7.5 (0.3)	7.2 (0.4)	7.4 (0.4)	8.1 (0.4)	7.8 (0.4)	8.3 (0.4)
Federal poverty threshold[4]										
Below 100%	9.9 (0.5)	10.2 (0.5)	9.4 (0.4)	9.5 (0.5)	10.4 (0.5)	11.8 (0.6)	10.8 (0.6)	11.0 (0.6)	11.7 (0.6)	11.2 (0.5)
100% to less than 250%	7.7 (0.3)	7.4 (0.3)	7.5 (0.3)	7.4 (0.3)	8.2 (0.3)	8.0 (0.4)	8.4 (0.4)	8.2 (0.4)	8.5 (0.4)	8.7 (0.3)
250% to less than 450%	6.8 (0.3)	6.4 (0.3)	6.4 (0.3)	6.6 (0.3)	7.3 (0.3)	7.3 (0.4)	7.0 (0.4)	7.6 (0.4)	7.5 (0.4)	8.2 (0.4)
450% or above	6.6 (0.3)	6.4 (0.3)	6.0 (0.3)	6.5 (0.3)	6.6 (0.3)	6.1 (0.3)	6.6 (0.3)	6.8 (0.4)	7.2 (0.4)	7.1 (0.3)

[1]Percentage is for the civilian noninstitutionalized population; except for age-specific estimates, percentages are age adjusted to the 2000 U.S. standard population.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race.

[4]The federal poverty threshold includes persons with reported and missing family income based on imputed income values using multiple imputation.

SOURCE: CDC/NCHS, National Health Interview Survey.

Table 2. Percentage with current asthma, by detailed age group and selected characteristics: United States, average annual 2008–2010

Characteristic	Total	0–17 years	18 years or over	0–4 years	5–14 years	15–34 years	15–19 years	20–24 years	25–34 years	35–64 years	65 years or over
	Percent (standard error)[1]										
Total	8.2 (0.1)	9.5 (0.2)	7.7 (0.1)	6.2 (0.4)	11.0 (0.3)	8.2 (0.2)	9.7 (0.4)	9.1 (0.5)	6.9 (0.3)	7.7 (0.2)	7.7 (0.3)
Sex											
Male	7.0 (0.1)	11.1 (0.3)	5.7 (0.2)	7.8 (0.5)	12.8 (0.5)	6.9 (0.3)	9.3 (0.7)	7.6 (0.6)	5.2 (0.4)	5.2 (0.2)	6.2 (0.4)
Female	9.2 (0.2)	7.9 (0.3)	9.7 (0.2)	4.4 (0.4)	9.0 (0.4)	9.5 (0.3)	10.2 (0.6)	10.7 (0.6)	8.5 (0.4)	10.1 (0.3)	8.8 (0.4)
Race[2]											
White	7.7 (0.1)	8.2 (0.3)	7.5 (0.1)	4.7 (0.4)	9.4 (0.3)	8.0 (0.2)	9.4 (0.5)	8.6 (0.5)	6.8 (0.3)	7.5 (0.2)	7.7 (0.3)
Male	6.5 (0.2)	9.7 (0.4)	5.5 (0.2)	6.1 (0.5)	11.3 (0.6)	6.6 (0.3)	6.9 (0.8)	7.0 (0.7)	5.2 (0.4)	5.1 (0.2)	6.2 (0.4)
Female	8.8 (0.2)	6.5 (0.3)	9.5 (0.2)	3.2 (0.4)	7.4 (0.4)	9.4 (0.4)	10.1 (0.7)	10.3 (0.9)	8.6 (0.5)	9.8 (0.3)	8.8 (0.4)
Black	11.2 (0.3)	16.0 (0.7)	9.2 (0.4)	12.3 (1.2)	18.8 (0.8)	9.7 (0.5)	12.1 (0.9)	11.8 (1.2)	7.3 (0.6)	9.5 (0.5)	8.4 (0.7)
Male	10.4 (0.5)	13.3 (1.0)	8.8 (0.5)	15.1 (1.8)	21.4 (1.3)	8.7 (0.8)	11.6 (1.4)	11.8 (2.1)	5.3 (0.9)	8.2 (0.6)	6.5 (1.0)
Female	11.8 (0.4)	13.7 (0.8)	11.1 (0.5)	9.4 (1.5)	16.1 (1.2)	10.7 (0.7)	12.6 (1.4)	11.9 (1.8)	9.0 (0.8)	12.1 (0.7)	9.8 (1.0)
Other races	8.0 (0.4)	9.5 (0.7)	7.3 (0.4)	7.7 (1.1)	11.3 (1.1)	7.4 (0.7)	8.0 (1.2)	9.0 (1.8)	6.4 (0.9)	7.3 (0.6)	6.4 (1.0)
Male	7.0 (0.5)	10.7 (0.9)	5.3 (0.5)	9.5 (1.8)	12.1 (1.4)	6.7 (0.9)	9.1 (1.9)	6.1 (1.8)	5.7 (1.0)	4.5 (0.6)	5.6 (1.2)
Female	8.9 (0.6)	8.3 (1.0)	9.1 (0.7)	5.8 (1.3)	10.4 (1.6)	8.2 (1.1)	6.8 (1.6)	11.6 (3.0)	7.1 (1.3)	9.9 (1.0)	7.1 (1.5)
Ethnicity[3]											
Hispanic or Latino	6.5 (0.2)	7.5 (0.4)	6.0 (0.3)	5.3 (0.6)	9.5 (0.6)	5.7 (0.3)	6.4 (0.6)	5.6 (0.7)	5.4 (0.5)	5.8 (0.3)	7.3 (0.9)
Male	5.9 (0.3)	8.7 (0.5)	4.4 (0.3)	5.6 (0.9)	10.8 (0.8)	4.9 (0.5)	6.7 (1.0)	4.1 (0.9)	4.5 (0.8)	3.8 (0.5)	5.2 (1.5)
Female	7.1 (0.3)	6.3 (0.5)	7.6 (0.4)	4.0 (0.7)	8.2 (0.8)	6.6 (0.5)	6.0 (0.9)	7.3 (1.3)	6.5 (0.6)	7.9 (0.5)	8.2 (1.0)
Puerto Rican	16.1 (1.0)	16.9 (1.8)	15.7 (1.3)	13.6 (2.7)	20.6 (2.6)	14.8 (1.7)	14.4 (3.6)	12.7 (3.2)	16.2 (3.1)	16.2 (1.7)	12.6 (2.9)
Male	15.6 (1.6)	21.5 (2.9)	12.2 (2.1)	17.1 (3.8)	27.0 (4.3)	15.1 (3.2)	*20.1 (7.8)	*13.2 (5.5)	*13.4 (5.4)	9.7 (2.5)	*10.6 (4.6)
Female	16.6 (1.1)	12.2 (1.8)	18.7 (1.7)	10.2 (2.9)	13.9 (2.7)	14.5 (1.7)	10.0 (2.1)	*12.4 (4.5)	18.0 (3.1)	22.0 (2.6)	14.4 (3.2)
Mexican	5.4 (0.2)	6.5 (0.5)	4.8 (0.3)	4.6 (0.7)	8.2 (0.7)	4.4 (0.3)	4.8 (0.6)	5.0 (0.8)	4.0 (0.5)	4.9 (0.4)	6.6 (1.3)
Male	4.7 (0.3)	7.2 (0.6)	3.3 (0.4)	5.7 (1.1)	8.6 (0.8)	3.3 (0.4)	5.1 (0.9)	3.1 (0.8)	2.5 (0.6)	3.3 (0.5)	*5.7 (1.8)
Female	6.2 (0.4)	5.7 (0.6)	6.5 (0.5)	3.4 (0.9)	7.7 (0.9)	5.7 (0.6)	4.5 (0.9)	7.0 (1.6)	5.7 (0.7)	6.7 (0.7)	7.4 (1.5)
Not Hispanic or Latino	8.5 (0.1)	10.0 (0.3)	8.0 (0.1)	6.4 (0.4)	11.4 (0.4)	8.8 (0.2)	10.5 (0.5)	9.9 (0.6)	7.2 (0.3)	8.0 (0.2)	7.7 (0.3)
Male	7.3 (0.2)	11.8 (0.4)	5.9 (0.2)	8.3 (0.6)	13.4 (0.6)	7.4 (0.3)	9.9 (0.6)	8.4 (0.8)	5.4 (0.4)	5.4 (0.2)	6.2 (0.4)
Female	9.6 (0.2)	8.2 (0.3)	10.0 (0.2)	4.5 (0.5)	9.2 (0.5)	10.1 (0.4)	11.2 (0.7)	11.3 (0.8)	9.0 (0.5)	10.4 (0.3)	8.8 (0.4)
Region											
Northeast	8.8 (0.3)	10.2 (0.5)	8.4 (0.3)	5.9 (0.7)	11.5 (0.6)	9.9 (0.5)	11.9 (1.0)	9.6 (1.4)	9.0 (0.7)	8.1 (0.4)	7.8 (0.5)
Midwest	8.7 (0.2)	10.4 (0.6)	8.1 (0.2)	7.4 (1.0)	11.5 (0.6)	9.2 (0.5)	10.9 (1.0)	10.4 (1.1)	7.7 (0.6)	8.2 (0.4)	6.6 (0.5)
South	7.6 (0.2)	10.0 (0.4)	6.8 (0.2)	6.9 (0.6)	11.7 (0.5)	6.9 (0.3)	8.5 (0.7)	8.1 (0.7)	5.4 (0.4)	6.9 (0.3)	7.4 (0.5)
West	9.0 (0.3)	7.4 (0.5)	8.2 (0.3)	4.1 (0.5)	8.9 (0.7)	7.9 (0.4)	8.4 (0.7)	9.1 (1.1)	7.0 (0.6)	8.3 (0.4)	9.0 (0.7)
Federal poverty threshold[4]											
Below 100%	11.2 (0.3)	12.4 (0.5)	10.6 (0.4)	9.6 (1.0)	14.4 (0.8)	9.1 (0.5)	9.3 (0.9)	9.6 (0.8)	8.5 (0.7)	12.7 (0.6)	9.9 (0.8)
100% to less than 250%	8.5 (0.2)	9.4 (0.4)	8.2 (0.2)	5.5 (0.5)	11.0 (0.6)	8.3 (0.4)	9.9 (1.0)	9.0 (0.6)	7.0 (0.5)	8.3 (0.4)	8.4 (0.5)
250% to less than 450%	7.7 (0.2)	8.4 (0.4)	7.5 (0.3)	4.9 (0.7)	9.7 (0.6)	7.7 (0.4)	10.0 (0.9)	7.9 (0.9)	6.7 (0.6)	7.7 (0.4)	7.4 (0.6)
450% or above	6.7 (0.2)	8.1 (0.4)	6.4 (0.2)	4.5 (0.6)	9.2 (0.6)	7.8 (0.5)	9.6 (1.0)	10.6 (1.6)	6.1 (0.5)	6.2 (0.2)	6.1 (0.6)

* Estimate is considered unreliable. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE over 50%.

[1]Percent is for the civilian noninstitutionalized population; crude percents are shown.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race.

[4]The federal poverty threshold includes persons with reported and missing family income based on imputed income values using multiple imputation.

SOURCE: CDC/NCHS, National Health Interview Survey.

Table 3. Percentage with one or more asthma attacks in the past 12 months, by year and selected characteristics: United States, 2001–2010

Characteristic	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
	Percent (standard error)[1]									
Total	4.3 (0.1)	4.3 (0.1)	3.9 (0.1)	4.1 (0.1)	4.2 (0.1)	4.2 (0.1)	4.2 (0.2)	4.3 (0.1)	4.3 (0.1)	4.6 (0.1)
Sex										
Male	3.5 (0.1)	3.5 (0.1)	3.1 (0.1)	3.6 (0.2)	3.4 (0.2)	3.4 (0.2)	3.3 (0.2)	3.7 (0.2)	3.5 (0.2)	3.6 (0.2)
Female	5.0 (0.2)	5.0 (0.2)	4.5 (0.2)	4.4 (0.2)	5.0 (0.2)	5.0 (0.2)	4.9 (0.2)	4.8 (0.2)	5.0 (0.2)	5.4 (0.2)
Race[2]										
White	4.2 (0.1)	4.1 (0.1)	3.7 (0.1)	3.9 (0.1)	4.1 (0.1)	4.2 (0.2)	4.0 (0.2)	4.0 (0.2)	4.1 (0.2)	4.3 (0.1)
Black	5.2 (0.3)	5.4 (0.3)	4.7 (0.3)	5.2 (0.3)	4.6 (0.3)	4.6 (0.3)	5.1 (0.4)	5.4 (0.4)	5.3 (0.4)	5.8 (0.4)
Other races	4.2 (0.3)	4.1 (0.4)	4.0 (0.5)	3.3 (0.3)	4.5 (0.5)	4.5 (0.5)	3.8 (0.4)	4.9 (0.6)	4.4 (0.5)	5.0 (0.5)
Ethnicity[3]										
Hispanic or Latino	3.4 (0.3)	2.9 (0.2)	3.3 (0.2)	3.1 (0.2)	3.5 (0.2)	3.6 (0.3)	4.1 (0.3)	3.2 (0.3)	3.5 (0.3)	4.0 (0.3)
Not Hispanic or Latino	4.5 (0.1)	4.5 (0.1)	4.0 (0.1)	4.3 (0.1)	4.3 (0.1)	4.4 (0.2)	4.2 (0.2)	4.5 (0.2)	4.5 (0.2)	4.7 (0.2)
Age										
Children (0–17)	5.7 (0.2)	5.8 (0.3)	5.5 (0.2)	5.5 (0.3)	5.2 (0.2)	5.6 (0.3)	5.2 (0.3)	5.6 (0.3)	5.5 (0.3)	5.7 (0.3)
Adults (18 or over)	3.8 (0.1)	3.8 (0.1)	3.3 (0.1)	3.6 (0.1)	3.9 (0.1)	3.8 (0.1)	3.8 (0.2)	3.8 (0.2)	3.9 (0.1)	4.2 (0.1)
Age group										
0–4 years	4.7 (0.4)	4.8 (0.4)	4.2 (0.4)	3.9 (0.4)	4.4 (0.5)	4.5 (0.5)	4.1 (0.5)	4.1 (0.5)	4.5 (0.5)	4.6 (0.6)
5–14 years	6.5 (0.4)	6.3 (0.4)	6.0 (0.4)	5.9 (0.3)	5.9 (0.3)	6.5 (0.5)	5.3 (0.4)	6.7 (0.5)	6.1 (0.4)	6.4 (0.4)
15–34 years	4.5 (0.2)	4.3 (0.2)	3.9 (0.2)	3.9 (0.2)	4.1 (0.2)	3.7 (0.2)	4.3 (0.3)	4.1 (0.3)	3.8 (0.2)	4.1 (0.2)
35–64 years	3.8 (0.2)	3.8 (0.2)	3.5 (0.2)	3.8 (0.2)	4.0 (0.2)	4.2 (0.2)	3.9 (0.2)	4.0 (0.2)	4.3 (0.2)	4.6 (0.2)
65 years or over	2.6 (0.2)	2.9 (0.3)	2.3 (0.2)	3.2 (0.3)	3.0 (0.3)	2.8 (0.3)	3.4 (0.3)	2.9 (0.3)	2.9 (0.3)	3.5 (0.3)
Region										
Northeast	4.7 (0.3)	4.7 (0.3)	4.3 (0.3)	4.4 (0.3)	4.9 (0.3)	4.5 (0.3)	4.5 (0.4)	4.1 (0.3)	4.7 (0.3)	4.2 (0.3)
Midwest	4.6 (0.2)	4.4 (0.2)	4.2 (0.3)	4.1 (0.2)	3.9 (0.2)	4.5 (0.3)	3.7 (0.3)	4.6 (0.3)	4.3 (0.3)	4.6 (0.3)
South	4.2 (0.2)	3.9 (0.2)	3.5 (0.2)	3.8 (0.2)	4.0 (0.2)	4.1 (0.2)	4.2 (0.3)	3.8 (0.2)	4.2 (0.3)	4.5 (0.2)
West	3.8 (0.2)	4.3 (0.3)	3.7 (0.2)	4.1 (0.2)	4.5 (0.3)	4.2 (0.3)	4.5 (0.3)	4.8 (0.3)	4.2 (0.3)	4.9 (0.3)
Federal poverty threshold[4]										
Below 100%	6.0 (0.4)	6.1 (0.4)	5.7 (0.3)	5.9 (0.4)	6.4 (0.4)	6.9 (0.4)	5.8 (0.4)	6.6 (0.5)	6.3 (0.4)	6.6 (0.4)
100% to less than 250%	4.5 (0.2)	4.4 (0.2)	4.1 (0.2)	4.4 (0.2)	4.4 (0.2)	4.5 (0.3)	4.7 (0.3)	4.4 (0.3)	4.7 (0.3)	4.7 (0.3)
250% to less than 450%	4.2 (0.2)	3.9 (0.2)	3.7 (0.2)	3.6 (0.2)	4.1 (0.2)	3.8 (0.3)	3.6 (0.3)	3.9 (0.3)	3.6 (0.3)	4.1 (0.3)
450% or above	3.8 (0.2)	3.8 (0.2)	3.2 (0.2)	3.5 (0.2)	3.5 (0.2)	3.2 (0.2)	3.6 (0.3)	3.7 (0.3)	4.0 (0.3)	3.9 (0.3)

[1]Percentage is for the civilian noninstitutionalized population; except for age-specific estimates, percentages are age adjusted to the 2000 U.S. standard population.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race.

[4]The federal poverty threshold includes persons with reported and missing family income based on imputed income values using multiple imputation.

SOURCE: CDC/NCHS, National Health Interview Survey.

Table 4. Percentage with one or more asthma attacks in the past 12 months, by detailed age group and selected characteristics: United States, average annual 2008–2010

Characteristic	Total	0–17 years	18 years or over	0–4 years	5–14 years	15–34 years	15–19 years	20–24 years	25–34 years	35–64 years	65 years or over
	Percent (standard error)[1]										
Total	4.4 (0.1)	5.6 (0.2)	4.0 (0.1)	4.4 (0.3)	6.4 (0.2)	4.0 (0.2)	4.3 (0.3)	4.3 (0.4)	3.7 (0.2)	4.3 (0.1)	3.1 (0.2)
Sex											
Male	3.6 (0.1)	6.5 (0.3)	2.6 (0.1)	5.6 (0.5)	7.5 (0.4)	3.0 (0.2)	3.8 (0.4)	2.9 (0.5)	2.6 (0.3)	2.7 (0.2)	2.4 (0.2)
Female	5.1 (0.1)	4.6 (0.2)	5.2 (0.1)	3.1 (0.4)	5.2 (0.3)	5.1 (0.2)	4.7 (0.4)	5.7 (0.6)	4.9 (0.3)	5.9 (0.2)	3.7 (0.2)
Race[2]											
White	4.1 (0.1)	4.9 (0.2)	3.9 (0.1)	3.6 (0.3)	5.6 (0.3)	3.9 (0.2)	4.0 (0.3)	4.2 (0.4)	3.6 (0.2)	4.2 (0.1)	3.1 (0.2)
Male	3.4 (0.1)	5.9 (0.3)	2.6 (0.1)	5.0 (0.5)	6.6 (0.4)	2.8 (0.2)	3.5 (0.5)	2.7 (0.5)	2.6 (0.3)	2.6 (0.2)	2.4 (0.3)
Female	4.8 (0.1)	3.8 (0.2)	5.1 (0.2)	2.2 (0.3)	4.2 (0.3)	5.1 (0.3)	4.5 (0.5)	5.8 (0.7)	5.0 (0.3)	5.7 (0.2)	3.7 (0.2)
Black	5.6 (0.2)	8.7 (0.4)	4.4 (0.2)	5.8 (0.9)	10.5 (0.7)	4.4 (0.4)	5.0 (0.6)	4.9 (0.9)	3.8 (0.4)	4.9 (0.4)	3.2 (0.4)
Male	4.9 (0.3)	9.6 (0.7)	2.8 (0.3)	7.7 (1.2)	11.8 (1.0)	3.5 (0.5)	4.7 (1.0)	*4.6 (1.6)	2.2 (0.6)	2.8 (0.4)	2.2 (0.5)
Female	6.2 (0.3)	7.7 (0.6)	5.7 (0.4)	5.9 (1.3)	9.2 (1.0)	5.2 (0.5)	5.4 (1.0)	5.2 (1.0)	5.1 (0.6)	6.6 (0.6)	3.8 (0.6)
Other races	4.9 (0.3)	6.5 (0.6)	4.2 (0.3)	6.6 (1.3)	8.7 (0.8)	4.2 (0.5)	5.3 (1.1)	*4.2 (1.4)	3.6 (0.6)	4.8 (0.5)	2.6 (0.8)
Male	4.1 (0.4)	6.9 (0.9)	2.8 (0.3)	7.5 (2.0)	6.6 (1.0)	3.3 (0.6)	5.3 (1.5)	*	3.0 (0.7)	3.0 (0.5)	*2.0 (0.7)
Female	5.7 (0.5)	6.1 (0.9)	5.5 (0.6)	5.7 (1.4)	8.7 (1.3)	5.0 (0.8)	5.4 (1.8)	*6.4 (2.5)	4.1 (0.9)	6.4 (0.8)	*3.5 (1.2)
Ethnicity[3]											
Hispanic or Latino	3.6 (0.2)	4.4 (0.3)	3.2 (0.2)	4.0 (0.5)	5.3 (0.4)	2.5 (0.2)	1.8 (0.3)	3.2 (0.5)	2.6 (0.3)	3.6 (0.2)	3.6 (0.6)
Male	3.1 (0.2)	5.2 (0.4)	2.0 (0.2)	5.1 (0.8)	5.9 (0.6)	1.9 (0.3)	1.9 (0.4)	*2.0 (0.6)	1.9 (0.5)	2.1 (0.3)	*2.9 (1.0)
Female	4.1 (0.2)	3.6 (0.4)	4.4 (0.3)	2.9 (0.6)	4.6 (0.5)	3.2 (0.3)	1.8 (0.4)	4.5 (0.9)	3.4 (0.4)	5.2 (0.4)	4.2 (0.7)
Puerto Rican	9.2 (0.9)	11.7 (1.6)	7.8 (0.9)	10.0 (2.5)	14.7 (2.2)	5.6 (0.8)	5.2 (1.5)	*7.7 (2.6)	4.7 (1.1)	9.9 (1.5)	*5.1 (1.7)
Male	6.9 (1.3)	15.9 (2.6)	5.1 (1.3)	12.9 (3.5)	20.4 (3.9)	*4.6 (1.5)	*8.2 (3.4)	*	*	*5.7 (2.2)	*
Female	9.4 (0.9)	7.4 (1.5)	10.3 (1.2)	*6.8 (2.7)	8.7 (2.1)	6.5 (1.1)	*	*7.4 (3.0)	8.4 (2.0)	13.8 (2.0)	*6.4 (2.2)
Mexican	2.9 (0.2)	3.4 (0.3)	2.6 (0.2)	3.3 (0.6)	4.0 (0.4)	2.1 (0.2)	1.0 (0.2)	2.8 (0.6)	2.3 (0.4)	2.9 (0.3)	3.7 (0.9)
Male	2.3 (0.2)	3.6 (0.4)	1.5 (0.3)	3.2 (0.6)	4.0 (0.5)	1.4 (0.3)	*1.1 (0.4)	*2.0 (0.8)	*1.3 (0.4)	1.5 (0.4)	*3.4 (1.6)
Female	3.6 (0.3)	3.2 (0.4)	3.8 (0.5)	2.8 (0.7)	3.9 (0.7)	2.9 (0.4)	1.0 (0.3)	3.6 (1.1)	3.5 (0.6)	4.4 (0.5)	3.9 (1.0)
Not Hispanic or Latino	4.5 (0.1)	5.9 (0.2)	4.1 (0.1)	4.5 (0.4)	6.7 (0.3)	4.4 (0.2)	4.8 (0.4)	4.6 (0.4)	4.0 (0.2)	4.4 (0.1)	3.1 (0.2)
Male	3.7 (0.1)	6.9 (0.3)	2.7 (0.1)	5.8 (0.6)	8.0 (0.4)	3.2 (0.2)	4.3 (0.5)	3.1 (0.5)	2.8 (0.3)	2.6 (0.2)	2.3 (0.2)
Female	5.2 (0.1)	4.9 (0.3)	5.3 (0.2)	5.2 (0.4)	5.4 (0.4)	5.5 (0.3)	5.4 (0.5)	6.0 (0.6)	5.3 (0.3)	5.9 (0.2)	3.7 (0.2)
Region											
Northeast	4.3 (0.2)	5.4 (0.4)	4.0 (0.2)	3.7 (0.6)	6.3 (0.6)	3.9 (0.4)	3.8 (0.6)	4.4 (1.0)	3.8 (0.5)	4.4 (0.3)	3.4 (0.4)
Midwest	4.5 (0.2)	6.2 (0.4)	3.9 (0.2)	5.1 (0.8)	6.8 (0.5)	4.9 (0.3)	5.3 (0.7)	5.0 (0.9)	4.6 (0.5)	4.0 (0.3)	2.4 (0.3)
South	4.1 (0.1)	6.0 (0.3)	3.5 (0.2)	5.0 (0.6)	7.0 (0.4)	3.2 (0.2)	3.9 (0.5)	3.4 (0.4)	2.8 (0.3)	4.0 (0.2)	3.2 (0.3)
West	4.6 (0.2)	4.8 (0.3)	4.6 (0.2)	3.3 (0.5)	5.4 (0.5)	4.4 (0.3)	4.0 (0.5)	5.0 (0.8)	4.2 (0.4)	5.0 (0.3)	3.6 (0.3)
Federal poverty threshold[4]											
Below 100%	6.4 (0.2)	7.4 (0.4)	5.9 (0.3)	6.1 (0.8)	8.3 (0.7)	4.9 (0.4)	4.7 (0.8)	4.8 (0.6)	5.1 (0.6)	7.6 (0.5)	4.8 (0.6)
100% to less than 250%	4.5 (0.2)	5.5 (0.3)	4.2 (0.2)	4.0 (0.5)	6.4 (0.4)	4.2 (0.3)	4.2 (0.6)	4.0 (0.5)	4.3 (0.4)	4.7 (0.3)	3.5 (0.3)
250& to less than 450%	3.9 (0.2)	4.8 (0.3)	3.6 (0.2)	3.8 (0.6)	5.6 (0.5)	3.4 (0.3)	3.9 (0.6)	3.5 (0.7)	3.2 (0.4)	4.0 (0.3)	2.8 (0.3)
450% or above	3.6 (0.1)	5.0 (0.3)	3.3 (0.2)	3.7 (0.6)	5.7 (0.5)	3.8 (0.3)	4.3 (0.6)	5.3 (1.1)	3.1 (0.4)	3.4 (0.2)	2.4 (0.3)

* Figure does not meet standards of reliability or precision. Estimates with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE over 50%.

[1]Percent is for the civilian noninstitutionalized population; crude percents are shown.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race.

[4]The federal poverty threshold includes persons with reported and missing family income based on imputed income values using multiple imputation.

SOURCE: CDC/NCHS, National Health Interview Survey.

Table 5. Percentage with one or more asthma attacks in the past 12 months among those with current asthma, by year and selected characteristics: United States, 2001–2010

Characteristic	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
	Percent (standard error)[1]									
Total	55.8 (1.0)	56.5 (1.0)	53.2 (1.1)	54.2 (1.1)	53.5 (1.0)	52.1 (1.2)	52.3 (1.4)	52.9 (1.3)	49.9 (1.2)	51.9 (1.1)
Sex										
Male	51.2 (1.6)	52.2 (1.7)	48.5 (1.8)	53.0 (1.7)	50.5 (1.7)	45.7 (2.0)	49.0 (2.2)	49.1 (2.0)	47.5 (1.9)	48.6 (1.8)
Female	58.5 (1.3)	58.6 (1.3)	55.0 (1.4)	54.6 (1.4)	54.8 (1.3)	56.4 (1.5)	54.6 (1.7)	55.3 (1.7)	51.6 (1.6)	53.1 (1.5)
Race[2]										
White	55.4 (1.2)	56.5 (1.2)	53.1 (1.3)	54.3 (1.3)	54.4 (1.2)	52.7 (1.4)	52.8 (1.7)	52.7 (1.5)	50.4 (1.4)	52.7 (1.3)
Black	59.2 (2.3)	57.0 (2.4)	50.4 (2.4)	55.4 (2.6)	46.8 (2.3)	47.3 (2.6)	51.1 (2.6)	51.9 (2.6)	47.2 (2.6)	47.3 (2.3)
Other races	56.4 (3.6)	61.1 (3.7)	57.2 (4.4)	47.8 (3.8)	60.7 (4.3)	52.0 (4.2)	50.3 (4.1)	58.0 (4.2)	56.1 (3.8)	57.4 (3.4)
Ethnicity[3]										
Hispanic or Latino	56.0 (2.9)	54.8 (2.9)	57.4 (2.8)	55.2 (2.9)	56.2 (2.7)	57.2 (3.0)	57.8 (2.6)	53.9 (3.3)	51.2 (2.8)	52.2 (2.8)
Not Hispanic or Latino	56.2 (1.1)	56.7 (1.1)	53.1 (1.2)	54.4 (1.2)	53.3 (1.1)	52.0 (1.3)	51.4 (1.6)	52.8 (1.3)	49.8 (1.4)	52.2 (1.2)
Age										
Children (0–17)	61.7 (1.8)	64.9 (1.8)	62.7 (1.7)	61.2 (1.7)	56.7 (1.8)	56.1 (2.2)	54.8 (2.3)	57.2 (2.2)	53.9 (2.2)	58.3 (1.9)
Adults (18 or over)	53.8 (1.2)	53.0 (1.2)	49.8 (1.3)	51.3 (1.3)	51.8 (1.2)	50.3 (1.5)	50.4 (1.7)	50.7 (1.5)	47.7 (1.4)	49.1 (1.3)
Age group										
0–4 years	70.7 (3.5)	77.3 (3.2)	69.7 (3.7)	65.1 (3.7)	60.8 (3.6)	69.9 (4.7)	59.0 (4.7)	64.4 (4.9)	66.2 (4.8)	68.3 (4.4)
5–14 years	63.4 (2.2)	63.6 (2.3)	62.7 (2.2)	61.5 (2.3)	58.5 (2.3)	56.0 (2.7)	53.9 (3.0)	59.4 (2.8)	51.7 (2.8)	58.2 (2.5)
15–34 years	54.0 (1.9)	53.4 (2.0)	51.4 (2.1)	51.8 (2.2)	52.2 (2.2)	45.6 (2.3)	49.1 (2.8)	48.5 (2.6)	46.6 (2.2)	45.4 (2.3)
35–64 years	55.5 (1.7)	54.0 (1.7)	52.5 (1.7)	54.7 (1.7)	55.3 (1.7)	56.4 (2.0)	54.1 (2.1)	54.3 (2.1)	51.9 (2.0)	54.1 (1.7)
65 years or over	43.2 (3.1)	49.9 (3.1)	38.8 (3.1)	42.5 (2.7)	37.7 (2.4)	37.5 (3.2)	44.3 (3.3)	40.5 (3.0)	36.9 (3.1)	41.6 (2.7)
Region										
Northeast	53.9 (2.2)	53.9 (2.2)	53.4 (2.2)	54.4 (2.5)	54.0 (2.2)	50.0 (2.7)	49.8 (2.9)	48.3 (2.8)	49.0 (2.5)	45.3 (2.8)
Midwest	58.4 (2.0)	58.2 (1.9)	51.9 (2.3)	53.7 (2.1)	48.2 (2.2)	52.4 (2.5)	43.3 (3.1)	52.3 (2.5)	47.6 (2.3)	50.6 (2.3)
South	56.1 (1.7)	55.6 (1.8)	53.5 (2.0)	54.9 (1.8)	54.1 (1.7)	51.8 (2.1)	56.8 (2.2)	52.2 (2.1)	52.0 (2.1)	52.4 (1.8)
West	54.1 (2.5)	59.0 (2.3)	55.0 (2.1)	53.1 (2.5)	58.3 (2.1)	54.9 (2.3)	57.1 (2.5)	56.6 (2.3)	50.5 (2.6)	56.5 (2.2)
Federal poverty threshold[4]										
Below 100%	58.7 (2.3)	56.9 (2.5)	57.2 (2.2)	60.8 (2.3)	59.4 (2.2)	55.3 (2.4)	54.2 (2.5)	59.2 (2.8)	51.4 (2.5)	55.9 (2.1)
100% to less than 250%	55.9 (2.0)	56.9 (2.2)	53.2 (2.0)	56.0 (2.0)	52.4 (2.1)	54.7 (2.3)	53.7 (2.5)	51.7 (2.7)	53.0 (2.4)	53.1 (2.1)
250% to less than 450%	57.9 (2.2)	58.0 (2.3)	55.6 (2.3)	50.9 (2.5)	54.9 (2.3)	51.0 (2.5)	49.9 (2.8)	51.2 (2.5)	45.9 (2.5)	48.2 (2.3)
450% or above	54.2 (2.2)	54.4 (2.2)	49.6 (2.4)	52.0 (2.3)	50.9 (2.1)	47.6 (2.7)	54.0 (2.4)	52.3 (2.6)	51.0 (2.4)	52.4 (2.5)

[1]Percent is for the civilian noninstitutionalized population with current asthma; except for age-specific estimates, percentages are age adjusted to the 2000 U.S. standard population. Estimates for 2001–2004 may not match those in the 2007 Surveillance Summary for this indicator (Table 10 in that publication) due to minor programming differences.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race.

[4]The federal poverty threshold includes persons with reported and missing family income based on imputed income values using multiple imputation.

SOURCE: CDC/NCHS, National Health Interview Survey.

Table 6. Percentage with one or more asthma attack in the past 12 months among those with current asthma, by detailed age group and selected characteristics: United States, average annual 2008–2010

Characteristic	Total	0–17 years	18 years or over	0–4 years	5–14 years	15–34 years	15–19 years	20–24 years	25–34 years	35–64 years	65 years or over
	Percent (standard error)[1]										
Total	51.2 (0.7)	56.5 (1.1)	49.1 (0.9)	66.3 (2.8)	56.4 (1.5)	46.8 (1.4)	42.1 (2.6)	45.0 (2.8)	51.4 (1.9)	53.5 (1.1)	39.7 (1.7)
Sex											
Male	49.2 (1.1)	56.3 (1.7)	44.4 (1.5)	67.5 (3.5)	56.3 (2.0)	41.1 (2.2)	39.6 (3.9)	35.7 (4.6)	46.5 (3.5)	49.4 (2.0)	37.2 (3.1)
Female	52.7 (0.9)	56.7 (1.8)	51.7 (1.1)	64.2 (4.2)	56.5 (2.3)	51.0 (1.8)	44.4 (3.2)	51.7 (3.7)	54.4 (2.4)	55.5 (1.4)	41.0 (2.1)
Race[2]											
White	51.0 (0.8)	56.9 (1.5)	49.1 (1.0)	66.5 (3.3)	57.2 (1.9)	46.9 (1.6)	41.0 (3.1)	45.9 (3.2)	51.7 (2.2)	53.4 (1.3)	39.7 (1.8)
Male	49.4 (1.3)	57.7 (2.0)	44.7 (1.7)	74.0 (4.0)	58.4 (2.5)	41.0 (2.7)	37.9 (4.7)	35.4 (5.3)	47.5 (4.1)	49.3 (2.3)	37.7 (3.4)
Female	52.2 (1.1)	55.5 (2.3)	51.6 (1.2)	59.6 (6.0)	55.4 (2.8)	51.2 (2.1)	43.7 (4.0)	53.5 (4.5)	54.3 (2.8)	55.5 (1.6)	40.9 (2.3)
Black	49.2 (1.3)	52.3 (1.9)	46.6 (1.9)	54.2 (4.9)	54.5 (2.8)	44.4 (2.8)	41.5 (3.9)	40.9 (5.7)	50.0 (4.6)	49.8 (2.7)	36.0 (4.1)
Male	46.4 (2.1)	51.4 (3.0)	40.0 (3.2)	49.4 (6.0)	53.5 (3.7)	39.7 (4.6)	40.8 (6.7)	39.3 (10.3)	38.5 (8.9)	43.7 (4.5)	32.5 (6.6)
Female	51.4 (1.7)	54.9 (3.1)	49.8 (2.3)	62.3 (7.8)	55.8 (4.5)	47.8 (3.3)	42.3 (5.3)	42.2 (7.0)	55.7 (4.8)	52.4 (3.1)	39.8 (5.3)
Other races	58.1 (2.4)	63.5 (3.7)	55.0 (3.0)	80.8 (7.8)	55.8 (4.8)	51.2 (4.6)	55.5 (8.5)	46.8 (11.2)	51.7 (5.9)	62.5 (3.9)	42.5 (8.5)
Male	54.2 (3.8)	60.0 (5.8)	48.8 (4.4)	79.6 (11.2)	48.7 (5.3)	45.4 (6.8)	51.4 (11.7)	*27.6 (13.3)	48.1 (9.2)	63.3 (6.1)	*33.7 (11.0)
Female	61.0 (2.8)	68.3 (5.1)	58.3 (3.7)	82.3 (7.2)	64.3 (7.1)	55.8 (6.1)	61.5 (11.7)	55.7 (13.4)	53.6 (8.2)	62.1 (4.7)	47.9 (11.1)
Ethnicity[3]											
Hispanic or Latino	52.2 (1.7)	54.8 (2.5)	50.4 (2.4)	66.9 (4.7)	52.8 (2.9)	41.3 (3.3)	26.1 (4.2)	51.2 (6.7)	45.2 (4.9)	58.5 (3.3)	46.6 (5.6)
Male	50.0 (2.6)	55.2 (3.2)	44.7 (4.3)	69.9 (6.7)	51.8 (3.6)	36.6 (5.7)	26.3 (6.2)	42.5 (11.0)	41.0 (9.1)	53.5 (5.6)	50.8 (11.4)
Female	54.0 (2.1)	54.2 (3.7)	54.0 (2.5)	61.6 (6.7)	54.1 (4.7)	45.3 (3.9)	25.9 (5.6)	55.6 (5.3)	48.7 (5.1)	61.1 (3.9)	47.5 (5.9)
Puerto Rican	54.1 (4.0)	67.0 (5.0)	47.1 (5.2)	70.9 (9.9)	68.2 (6.4)	35.2 (5.8)	*29.7 (10.8)	56.7 (11.8)	28.5 (7.2)	57.8 (7.0)	38.0 (10.8)
Male	52.8 (6.0)	69.3 (8.9)	36.6 (9.1)	73.8 (13.8)	69.6 (8.5)	*24.0 (9.3)	*	*52.7 (20.3)	*	56.5 (13.8)	*
Female	55.3 (4.5)	62.5 (7.7)	53.1 (5.5)	66.1 (16.4)	65.4 (5.4)	45.1 (8.9)	*	59.6 (14.9)	44.8 (7.6)	58.3 (8.0)	44.1 (12.2)
Mexican	51.0 (2.2)	56.5 (3.3)	51.3 (3.0)	66.2 (5.4)	47.2 (3.7)	44.7 (4.4)	20.5 (4.8)	52.7 (9.2)	54.3 (6.3)	55.5 (4.8)	52.7 (8.2)
Male	47.8 (3.4)	48.9 (4.3)	48.0 (5.3)	65.1 (9.2)	44.7 (4.3)	41.1 (6.5)	*21.0 (7.0)	57.2 (13.1)	50.3 (10.8)	45.6 (8.3)	59.7 (13.8)
Female	53.7 (2.7)	52.6 (4.8)	54.2 (3.4)	65.0 (7.4)	50.3 (6.0)	47.1 (5.7)	20.0 (5.9)	50.4 (11.2)	56.5 (7.8)	60.7 (5.3)	48.2 (11.2)
Not Hispanic or Latino	51.1 (0.8)	56.8 (1.3)	49.0 (0.9)	66.2 (3.3)	57.2 (1.7)	47.6 (1.5)	44.3 (2.9)	44.3 (3.0)	52.6 (2.1)	52.9 (1.2)	39.1 (1.7)
Male	49.1 (1.2)	56.6 (1.9)	44.3 (1.5)	66.8 (4.1)	57.3 (2.3)	41.9 (2.4)	41.6 (4.4)	34.9 (4.9)	47.8 (3.7)	49.0 (2.2)	36.3 (3.1)
Female	52.6 (1.0)	57.2 (2.0)	51.5 (1.1)	64.9 (5.1)	57.0 (2.6)	51.8 (1.9)	46.7 (3.5)	51.0 (3.9)	55.4 (2.8)	54.9 (1.5)	40.6 (2.2)
Region											
Northeast	47.3 (1.7)	50.2 (2.8)	46.2 (2.1)	59.0 (7.5)	52.3 (3.8)	37.9 (3.5)	30.7 (5.6)	45.2 (7.8)	39.2 (4.6)	52.5 (2.5)	41.7 (3.7)
Midwest	50.2 (1.3)	57.5 (2.2)	47.1 (1.7)	67.8 (5.0)	58.3 (2.9)	51.1 (2.7)	48.4 (5.8)	47.2 (5.9)	56.0 (3.8)	47.9 (2.2)	34.5 (3.5)
South	52.2 (1.2)	57.6 (1.8)	49.7 (1.5)	65.0 (4.0)	58.1 (2.4)	44.9 (2.4)	43.2 (3.9)	39.6 (4.2)	50.4 (3.6)	54.9 (2.1)	42.6 (3.0)
West	54.1 (1.6)	58.6 (2.6)	52.7 (1.8)	73.1 (7.3)	56.8 (3.3)	51.4 (2.9)	43.0 (4.8)	49.7 (5.6)	57.5 (3.2)	58.0 (2.4)	38.5 (3.1)
Federal poverty threshold[4]											
Below 100%	55.1 (1.4)	56.9 (2.4)	54.0 (1.7)	58.9 (5.3)	55.7 (3.2)	51.8 (2.5)	47.2 (4.9)	48.5 (4.6)	58.4 (4.1)	58.1 (2.4)	47.5 (4.6)
100% to less than 250%	51.4 (1.3)	56.3 (2.1)	49.2 (1.5)	67.0 (4.2)	56.0 (2.7)	48.0 (2.5)	41.1 (4.9)	41.4 (4.9)	58.7 (3.8)	54.7 (2.4)	40.2 (2.6)
250% to less than 450%	47.9 (1.5)	55.1 (2.5)	45.5 (1.8)	73.0 (5.3)	55.7 (3.0)	42.4 (3.1)	39.0 (5.0)	43.4 (5.9)	44.3 (4.1)	49.6 (2.4)	37.2 (3.3)
450% or above	51.3 (1.5)	57.8 (2.8)	49.5 (1.7)	75.6 (8.2)	58.9 (3.6)	45.6 (3.2)	42.7 (5.4)	48.6 (7.5)	46.0 (4.1)	53.4 (2.0)	37.7 (4.3)

* Figure does not meet standards of reliability or precision. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE over 50%.

[1]Percentage is for the civilian noninstitutionalized population with current asthma; crude percents are shown.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race.

[4]The federal poverty threshold includes persons with reported and missing family income based on imputed income values using multiple imputation.

SOURCE: CDC/NCHS, National Health Interview Survey.

Table 7. Rate of physician office visits with asthma as the first-listed diagnosis per 10,000 population, by year and selected patient characteristics: United States, 2001–2009

Characteristic	2001	2002	2003	2004	2005	2006	2007	2008	2009
	Rate (standard error)[1]								
Total	409.7 (44.0)	457.4 (40.2)	448.5 (40.6)	470.2 (50.9)	440.3 (45.2)	361.3 (39.8)	475.3 (45.5)	298.6 (30.3)	351.2 (32.9)
Sex									
Male	405.5 (64.2)	344.9 (43.5)	409.9 (52.7)	418.9 (58.3)	449.6 (58.4)	282.2 (41.5)	487.8 (57.5)	262.1 (38.3)	309.4 (36.8)
Female	405.9 (54.4)	549.3 (57.9)	484.1 (54.8)	514.7 (68.0)	423.2 (53.9)	429.4 (57.9)	452.9 (59.4)	325.8 (43.2)	386.2 (48.7)
Race[2]									
White	434.2 (49.3)	473.2 (45.1)	468.8 (45.0)	476.6 (57.7)	445.7 (47.8)	358.7 (40.6)	481.5 (51.1)	285.9 (31.5)	354.1 (34.5)
Black	*299.5 (98.2)	368.7 (72.4)	349.2 (89.8)	477.1 (109.5)	411.5 (102.0)	314.5 (92.3)	594.9 (119.5)	342.4 (73.4)	410.7 (111.9)
Other races	*294.1 (93.4)	*576.7 (386.6)	*469.6 (171.7)	*420.8 (146.9)	*463.0 (166.9)	*383.7 (119.7)	*227.6 (87.4)	*370.0 (162.2)	123.7 (60.2)
Ethnicity[3]									
Hispanic or Latino	...	...	460.4 (130.9)	422.9 (99.4)	530.2 (169.7)	287.8 (93.2)	457.9 (93.1)	446.4 (84.8)	426.8 (81.0)
Not Hispanic or Latino	...	...	456.9 (43.5)	472.3 (53.6)	414.0 (38.8)	371.8 (40.0)	475.6 (52.2)	271.2 (31.5)	343.2 (35.0)
Age									
Children (0–17)	605.7 (129.8)	597.7 (91.7)	738.9 (133.0)	895.1 (208.9)	743.7 (176.4)	465.9 (102.1)	902.3 (178.6)	481.8 (95.5)	884.2 (97.3)
Adults (18 or over)	337.5 (65.6)	404.8 (76.4)	350.3 (60.0)	328.4 (57.0)	338.3 (54.6)	325.4 (75.6)	323.5 (56.8)	237.5 (37.0)	310.8 (48.9)
Age group									
0–4 years	1,064.9 (303.7)	735.0 (161.6)	1,092.5 (296.0)	1,095.2 (293.4)	938.7 (236.3)	308.6 (102.8)	1,167.4 (249.7)	646.3 (169.9)	444.1 (129.2)
5–14 years	468.5 (105.1)	631.8 (116.5)	667.2 (130.4)	817.8 (222.4)	677.7 (192.9)	606.4 (148.7)	965.5 (196.3)	445.6 (104.2)	516.2 (108.9)
15–34 years	251.2 (59.8)	250.4 (50.7)	218.4 (49.6)	332.0 (76.5)	286.5 (71.1)	186.7 (40.8)	*290.7 (92.0)	146.4 (34.7)	221.2 (41.9)
35–64 years	292.8 (69.2)	355.3 (76.0)	389.6 (77.5)	361.5 (73.8)	392.5 (62.8)	341.8 (75.9)	315.7 (60.4)	261.2 (47.6)	329.7 (56.5)
65 years or over	675.9 (192.5)	870.4 (222.3)	545.6 (148.9)	389.7 (108.1)	387.7 (112.1)	*566.2 (261.1)	408.9 (115.1)	393.3 (134.8)	468.0 (164.5)
Region									
Northeast	431.1 (96.7)	625.4 (114.2)	377.9 (83.3)	689.7 (170.4)	320.5 (67.6)	600.7 (136.5)	604.9 (153.7)	509.2 (108.2)	454.0 (81.6)
Midwest	353.0 (101.5)	277.5 (56.9)	453.6 (78.0)	455.3 (71.3)	451.3 (81.2)	271.5 (54.3)	341.6 (64.6)	210.0 (39.0)	360.4 (62.2)
South	392.1 (68.2)	455.1 (71.2)	428.0 (70.5)	455.5 (81.7)	528.5 (99.5)	385.3 (71.9)	610.1 (84.1)	245.2 (39.2)	370.4 (64.0)
West	483.6 (98.1)	524.7 (83.8)	551.9 (96.9)	324.4 (96.6)	390.0 (74.8)	218.7 (45.1)	302.6 (64.9)	306.7 (68.7)	224.7 (4.5)

* Figure does not meet standards of reliability or precision. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE over 50%.

... Category not applicable.

[1]Except for age-specific estimates, rates are age adjusted to the 2000 U.S. standard population.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race. Ethnicity was not imputed in 2001 or 2002. It is not shown for those years due to a high rate of missing data.

SOURCE: CDC/NCHS, National Ambulatory Medical Care Survey.

Table 8. Rate of physician office visits with asthma as the first-listed diagnosis per 100 persons with current asthma, by year and selected patient characteristics: United States, 2001–2009

	2001	2002	2003	2004	2005	2006	2007	2008	2009
	Rate (standard error)[1]								
Total	55.6 (9.7)	63.4 (9.6)	64.8 (10.2)	66.2 (11.3)	57.7 (10.1)	46.3 (8.3)	60.6 (10.3)	38.2 (5.0)	43.0 (5.9)
Sex									
Male	63.9 (14.3)	56.1 (8.7)	69.2 (11.1)	67.0 (11.3)	71.9 (16.7)	39.8 (6.8)	74.6 (12.7)	36.9 (7.0)	43.9 (6.9)
Female	49.6 (9.3)	68.7 (12.0)	61.7 (11.1)	65.7 (13.1)	47.6 (7.5)	51.4 (11.7)	50.7 (11.0)	39.2 (6.2)	42.4 (7.1)
Race[2]									
White	60.9 (10.9)	69.1 (11.2)	71.0 (12.0)	69.3 (12.5)	60.0 (10.5)	48.3 (9.4)	63.9 (12.2)	39.4 (5.7)	46.5 (6.9)
Black	*32.9 (11.4)	42.4 (11.4)	37.1 (9.7)	53.3 (12.8)	*45.6 (17.4)	*38.6 (15.1)	61.5 (14.4)	34.5 (9.1)	39.7 (11.1)
Other races	*45.7 (16.0)	*51.3 (22.4)	*58.8 (23.2)	*59.8 (21.3)	*58.6 (25.4)	*38.6 (12.8)	*22.2 (8.3)	*	*14.0 (6.9)
Ethnicity[3]									
Hispanic or Latino	...	...	*74.8 (24.5)	83.8 (24.8)	*	*44.7 (15.8)	75.9 (18.2)	77.6 (17.5)	64.8 (15.0)
Not Hispanic or Latino	...	...	63.6 (10.6)	63.0 (11.6)	52.5 (6.8)	46.5 (8.8)	58.2 (11.5)	33.0 (4.8)	40.0 (5.9)
Age									
Children (0–17)	69.6 (15.1)	71.9 (11.4)	86.8 (16.0)	105.7 (25.0)	83.6 (20.1)	50.2 (11.2)	99.3 (20.2)	51.2 (10.4)	51.9 (10.4)
Adults (18 or over)	49.3 (9.7)	59.7 (11.4)	54.8 (9.5)	49.2 (8.6)	46.9 (7.7)	44.6 (10.5)	44.6 (6.0)	32.7 (5.2)	39.4 (6.5)
Age group									
0–4 years	186.4 (55.1)	123.4 (29.2)	184.2 (52.1)	195.5 (54.7)	139.0 (36.7)	*53.4 (18.6)	171.8 (40.1)	105.9 (30.0)	*71.0 (21.6)
5–14 years	47.5 (10.9)	69.1 (13.2)	70.7 (14.3)	90.0 (24.8)	69.2 (20.0)	55.9 (14.1)	103.0 (21.9)	40.8 (9.8)	46.0 (10.0)
15–34 years	31.6 (7.6)	33.2 (6.9)	30.4 (7.0)	47.0 (11.0)	38.1 (9.6)	25.8 (5.4)	*34.3 (11.2)	18.2 (4.4)	28.5 (5.6)
35–64 years	44.0 (10.5)	62.1 (11.3)	61.1 (12.4)	53.4 (11.0)	55.1 (9.0)	46.9 (10.6)	45.6 (9.0)	36.9 (6.9)	41.8 (7.4)
65 years or over	113.5 (33.1)	146.8 (39.1)	94.8 (26.5)	58.5 (15.7)	50.9 (15.0)	*80.9 (37.8)	54.7 (15.8)	*54.8 (19.1)	*61.0 (21.9)
Region									
Northeast	51.9 (14.6)	74.9 (20.0)	*47.5 (14.4)	*93.1 (43.0)	*37.3 (11.4)	*67.5 (21.8)	*	63.7 (17.1)	*5.0 (16.1)
Midwest	*45.9 (21.5)	37.6 (9.7)	58.3 (15.6)	61.3 (13.7)	59.3 (14.7)	31.4 (7.7)	*26.8 (12.0)	23.2 (5.9)	39.8 (8.8)
South	*54.2 (16.6)	*67.3 (20.9)	*69.6 (21.3)	68.0 (17.1)	*71.4 (25.6)	*52.5 (18.6)	82.7 (17.4)	36.5 (7.6)	49.5 (12.5)
West	*75.6 (25.0)	76.0 (21.1)	*84.4 (28.9)	*46.1 (16.1)	53.5 (12.9)	31.2 (8.1)	42.2 (11.8)	*38.0 (12.0)	29.3 (8.2)

* Figure does not meet standards of reliability or precision. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE greater than 50%.

... Category not applicable.

[1]Crude rates are shown.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race. Ethnicity was not imputed in 2001 or 2002. It is not shown for those years due to a high rate of missing data.

SOURCE: CDC/NCHS, National Ambulatory Medical Care Survey.

Table 9. Rate of hospital outpatient department visits with asthma as the first-listed diagnosis per 10,000 population, by year and selected patient characteristics: United States, 2001–2009

	2001	2002	2003	2004	2005	2006	2007	2008	2009
	Rate (standard error)[1]								
Total	46.4 (5.2)	43.5 (4.8)	52.9 (6.2)	36.6 (3.6)	44.6 (4.1)	37.5 (3.5)	48.0 (7.6)	55.8 (6.5)	39.6 (5.8)
Sex									
Male	34.9 (4.9)	41.4 (5.9)	43.4 (7.3)	32.2 (5.1)	36.6 (5.3)	32.3 (4.1)	44.5 (7.9)	48.0 (7.2)	41.5 (7.4)
Female	56.3 (8.4)	45.2 (5.9)	61.6 (7.2)	39.8 (4.5)	51.9 (6.3)	41.9 (5.3)	51.4 (9.0)	62.9 (8.4)	37.1 (5.7)
Race[2]									
White	43.9 (5.8)	32.9 (4.4)	36.7 (5.2)	24.2 (2.9)	35.8 (3.8)	32.3 (3.8)	35.6 (6.2)	45.8 (6.5)	33.6 (4.5)
Black	76.1 (10.4)	115.7 (16.7)	152.1 (24.0)	116.4 (19.3)	111.3 (19.8)	65.0 (10.6)	128.4 (27.4)	105.2 (18.9)	*69.1 (20.8)
Other races	*15.4 (4.7)	*20.0 (7.4)	52.6 (15.7)	*46.1 (20.0)	*31.2 (10.2)	*42.2 (13.6)	*14.7 (4.8)	52.5 (13.6)	*43.9 (14.4)
Ethnicity[3]									
Hispanic or Latino	...	...	117.7 (29.8)	59.5 (14.5)	70.6 (14.0)	41.4 (8.6)	85.0 (20.0)	73.7 (17.9)	26.6 (6.1)
Not Hispanic or Latino	...	...	47.8 (6.6)	35.2 (3.9)	40.5 (4.2)	36.2 (3.8)	41.5 (6.4)	55.4 (7.2)	44.5 (7.3)
Age									
Children (0–17)	78.5 (18.6)	87.0 (17.3)	102.8 (27.6)	63.2 (10.7)	85.6 (14.5)	64.7 (9.7)	*114.2 (39.5)	113.8 (27.9)	103.9 (27.9)
Adults (18 or over)	35.1 (8.4)	28.7 (4.9)	35.8 (9.5)	27.6 (5.1)	31.0 (5.4)	28.0 (5.0)	25.4 (6.0)	*36.5 (7.7)	17.8 (2.6)
Age group									
0–4 years	128.8 (33.9)	110.9 (25.2)	127.7 (38.3)	105.1 (25.9)	121.7 (29.0)	67.4 (14.8)	*154.4 (53.2)	140.2 (48.1)	123.1 (33.9)
5–14 years	71.5 (18.2)	95.1 (23.0)	106.6 (28.9)	48.4 (9.8)	77.3 (13.5)	73.1 (12.3)	*110.0 (42.3)	117.6 (27.0)	*105.2 (33.4)
15–34 years	31.8 (8.6)	35.4 (8.1)	37.7 (7.6)	28.3 (5.9)	31.6 (5.6)	26.5 (5.6)	33.5 (8.1)	*46.7 (12.2)	32.3 (6.2)
35–64 years	34.9 (9.4)	24.7 (4.9)	*35.6 (11.6)	28.1 (6.0)	36.1 (8.3)	30.8 (6.4)	25.3 (6.8)	30.5 (6.2)	16.1 (3.7)
65 years or over	*38.1 (15.7)	*23.4 (9.1)	*36.9 (14.0)	*29.9 (11.0)	*21.6 (8.0)	*24.4 (9.9)	*21.0 (10.1)	*40.9 (17.6)	*
Region									
Northeast	54.1 (9.4)	62.0 (9.0)	105.3 (17.4)	60.8 (9.1)	85.3 (12.8)	52.5 (9.2)	74.7 (13.6)	76.8 (13.6)	76.8 (14.1)
Midwest	67.0 (12.8)	45.2 (10.3)	32.3 (7.2)	42.2 (8.0)	53.4 (9.3)	57.0 (9.5)	31.3 (6.0)	82.3 (16.3)	50.3 (10.0)
South	44.8 (10.3)	44.8 (9.8)	54.2 (12.6)	24.3 (5.1)	26.7 (5.8)	26.5 (5.2)	*53.6 (16.7)	54.3 (12.9)	*32.3 (12.9)
West	19.2 (5.0)	23.4 (5.5)	25.5 (7.3)	31.1 (7.7)	30.0 (6.4)	24.2 (5.1)	*36.0 (15.4)	17.8 (4.7)	*14.7 (4.9)

* Figure does not meet standards of reliability or precision. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE greater than 50%.

... Category not applicable.

[1]Except for age-specific estimates, rates are age adjusted to the 2000 U.S. standard population.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race. Ethnicity was not imputed in 2001 or 2002. It is not shown for those years due to a high rate of missing data.

SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

Table 10. Rate of hospital outpatient department visits with asthma as the first-listed diagnosis per 100 persons with current asthma, by year and selected patient characteristics: United States, 2001–2009

Characteristic	2001	2002	2003	2004	2005	2006	2007	2008	2009
	Rate (standard error)[1]								
Total	6.3 (1.1)	6.1 (0.9)	7.6 (1.5)	5.1 (0.7)	5.9 (0.8)	4.8 (0.7)	6.2 (1.5)	7.1 (1.2)	4.8 (0.9)
Sex									
Male	5.7 (1.0)	7.0 (1.3)	7.5 (1.8)	5.2 (1.0)	5.9 (1.1)	4.7 (0.7)	6.8 (1.8)	6.8 (1.3)	5.9 (1.3)
Female	6.8 (1.5)	5.5 (0.9)	7.7 (1.6)	5.0 (0.9)	5.9 (0.9)	4.8 (0.8)	5.7 (1.4)	7.4 (1.3)	3.9 (0.7)
Race[2]									
White	6.1 (1.4)	4.8 (1.0)	5.5 (1.2)	3.5 (0.6)	4.8 (0.6)	4.2 (0.7)	4.7 (1.3)	6.2 (1.2)	4.3 (0.7)
Black	10.0 (1.8)	14.0 (2.5)	16.1 (4.5)	12.9 (2.6)	11.8 (2.9)	8.0 (1.7)	*14.5 (4.6)	12.1 (3.2)	*7.1 (2.6)
Other races	*2.0 (0.7)	*2.9 (1.3)	*6.8 (2.6)	*6.4 (2.8)	*4.3 (1.5)	*4.2 (1.5)	*1.7 (0.7)	*5.2 (1.9)	*4.7 (1.7)
Ethnicity[3]									
Hispanic or Latino	...	...	*16.8 (8.0)	9.7 (2.8)	12.0 (3.3)	7.6 (1.9)	*13.6 (5.1)	11.9 (3.0)	4.4 (1.2)
Not Hispanic or Latino	...	...	6.5 (1.3)	4.6 (0.7)	5.0 (0.7)	4.4 (0.7)	5.0 (1.1)	6.5 (1.2)	4.8 (1.0)
Age									
Children (0–17)	9.0 (2.2)	10.5 (2.1)	12.1 (3.3)	7.5 (1.3)	9.6 (1.7)	7.0 (1.1)	*12.5 (4.4)	12.1 (3.0)	10.8 (2.9)
Adults (18 or over)	5.1 (1.2)	4.2 (0.7)	5.6 (1.5)	4.1 (0.8)	4.3 (0.8)	3.8 (0.7)	3.5 (0.8)	5.0 (1.1)	2.3 (0.4)
Age group									
0–4 years	22.5 (6.2)	18.8 (4.5)	*21.5 (6.7)	18.8 (4.9)	16.0 (4.5)	11.7 (2.8)	*22.7 (8.1)	*23.0 (8.3)	19.6 (5.7)
5–14 years	7.2 (1.9)	10.4 (2.6)	11.3 (3.1)	5.3 (1.1)	7.9 (1.4)	6.7 (1.2)	*11.7 (4.6)	10.8 (2.6)	*9.4 (3.0)
15–34 years	4.0 (1.1)	4.7 (1.1)	5.2 (1.1)	4.0 (0.9)	4.2 (0.8)	3.4 (0.7)	4.0 (1.0)	5.8 (1.5)	4.2 (0.8)
35–64 years	5.2 (1.4)	3.6 (0.7)	*5.6 (1.8)	4.1 (0.9)	5.1 (1.2)	4.2 (0.9)	3.7 (1.0)	4.3 (0.9)	2.0 (0.5)
65 years or over	*6.4 (2.7)	*4.0 (1.6)	*6.4 (2.5)	*4.3 (1.6)	*2.8 (1.0)	*3.5 (1.4)	*2.8 (1.4)	*5.7 (2.5)	*
Region									
Northeast	*6.6 (2.0)	7.5 (1.4)	*13.6 (5.0)	8.3 (2.2)	10.1 (2.6)	5.9 (1.4)	*9.0 (3.3)	9.2 (1.8)	7.8 (1.8)
Midwest	*8.8 (3.1)	6.2 (1.5)	4.1 (1.0)	5.6 (1.5)	6.9 (1.6)	6.6 (1.8)	3.7 (0.9)	*9.3 (2.8)	6.5 (1.4)
South	*6.1 (2.0)	*6.6 (2.2)	*8.7 (3.2)	3.5 (0.9)	3.6 (1.0)	3.5 (0.9)	*7.1 (3.2)	*7.8 (2.7)	*
West	*3.2 (1.1)	*3.5 (1.1)	*4.0 (1.6)	*4.4 (1.8)	*4.2 (1.5)	*2.5 (1.2)	*	*2.3 (0.8)	*1.9 (0.7)

* Figure does not meet standards of reliability or precision. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE over 50%.

... Category not applicable.

[1]Crude rates are shown.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race. Ethnicity was not imputed in 2001 or 2002. It is not shown for those years due to a high rate of missing data.

SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

Table 11. Rate of physician office visits with asthma as the first-listed diagnosis per 100 persons with current asthma, by detailed age group and selected patient characteristics: United States, average annual 2007–2009

Characteristic	Total	0–17 years	18 years or over	0–4 years	5–14 years	15–34 years	35–64 years	65 years or over
	Rate (standard error)[1]							
Total	47.1 (4.8)	66.9 (8.5)	38.9 (4.4)	117.2 (18.5)	61.2 (8.5)	27.2 (4.7)	41.4 (4.9)	57.0 (12.2)
Sex								
Male	51.1 (5.4)	74.9 (9.5)	35.0 (4.6)	118.6 (19.7)	72.5 (10.6)	24.3 (4.8)	42.5 (7.1)	39.6 (10.8)
Female	44.1 (5.6)	55.5 (11.0)	41.1 (5.5)	114.5 (35.7)	44.5 (9.5)	29.4 (6.1)	40.9 (5.7)	66.5 (16.6)
Race[2]								
White	49.9 (5.7)	80.5 (11.3)	39.7 (4.9)	156.9 (28.9)	72.0 (10.8)	30.6 (6.1)	42.0 (5.3)	54.7 (13.4)
Male	55.4 (6.3)	90.2 (12.4)	36.6 (5.0)	158.1 (30.0)	85.2 (13.4)	27.5 (5.9)	42.3 (7.1)	44.3 (12.3)
Female	45.8 (6.7)	66.0 (15.1)	41.6 (6.3)	*154.8 (56.9)	51.2 (12.8)	32.8 (7.9)	41.8 (6.3)	*60.5 (18.5)
Black	44.9 (7.0)	52.3 (10.7)	38.9 (8.5)	89.5 (26.4)	47.6 (13.6)	*19.0 (5.8)	43.4 (12.8)	*73.4 (36.6)
Male	48.0 (10.4)	55.3 (12.8)	*37.3 (17.0)	*90.0 (29.4)	*51.2 (15.5)	*21.2 (10.0)	*	*
Female	42.6 (7.9)	48.3 (14.0)	39.6 (9.0)	*88.8 (44.0)	*42.9 (16.1)	*17.6 (6.2)	39.7 (11.1)	*
Other races	*23.5 (7.5)	*16.4 (6.1)	*27.9 (11.6)	*	*25.4 (10.0)	*	*	68.7 (34.0)
Male	*16.3 (6.1)	*22.6 (10.1)	*	*	*37.8 (17.2)	*	*	*
Female	*29.3 (12.6)	*	38.0 (17.8)	*	*	*	*	*
Ethnicity[3]								
Hispanic or Latino	72.6 (9.9)	95.6 (17.8)	55.5 (10.7)	183.1 (45.2)	67.3 (15.4)	*35.4 (11.1)	72.0 (15.7)	*71.4 (29.3)
Male	76.7 (14.7)	101.2 (23.7)	47.4 (14.1)	191.0 (56.1)	73.6 (21.3)	*33.9 (15.2)	*67.3 (25.1)	*
Female	69.1 (11.5)	87.5 (19.1)	60.1 (14.4)	169.1 (57.5)	59.0 (18.7)	*36.6 (15.9)	74.3 (19.6)	*99.3 (45.9)
Not Hispanic or Latino	43.5 (5.1)	60.6 (9.3)	37.0 (4.7)	97.6 (18.4)	59.8 (9.7)	26.1 (5.2)	38.1 (5.0)	55.9 (12.5)
Male	47.2 (5.5)	69.2 (9.7)	33.6 (4.9)	97.2 (18.6)	72.3 (11.9)	23.0 (5.1)	39.9 (7.3)	40.6 (11.6)
Female	40.8 (6.1)	48.3 (12.5)	39.0 (5.7)	98.2 (40.3)	40.8 (10.8)	28.4 (6.6)	37.2 (5.7)	64.2 (17.0)
Region								
Northeast	61.4 (16.5)	*76.3 (27.9)	55.2 (13.9)	*132.9 (45.6)	*70.0 (25.3)	*	49.5 (13.3)	*112.2 (40.7)
Midwest	33.9 (6.0)	38.5 (8.0)	32.0 (6.8)	*50.3 (23.1)	38.0 (8.3)	19.9 (4.8)	41.3 (10.4)	*35.5 (13.3)
South	56.6 (8.5)	86.3 (15.5)	42.2 (8.1)	156.2 (35.0)	72.4 (16.0)	35.9 (7.1)	39.7 (7.2)	*63.6 (26.2)
West	36.3 (7.6)	58.8 (15.3)	29.1 (7.5)	*98.4 (34.0)	62.5 (18.3)	*14.1 (5.3)	37.2 (9.8)	*30.1 (12.8)

* Figure does not meet standards of reliability or precision. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE over 50%.

[1]Crude estimates are shown.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race.

SOURCE: CDC/NCHS, National Ambulatory Medical Care Survey.

Table 12. Rate of hospital outpatient department visits with asthma as the first-listed diagnosis per 100 persons with current asthma, by detailed age group and selected patient characteristics: United States, average annual 2007–2009

Characteristic	Total	0–17 years	18 years or over	0–4 years	5–14 years	15–34 years	35–64 years	65 years or over
				Rate (standard error)[1]				
Total	6.0 (0.8)	11.8 (2.4)	3.6 (0.5)	21.8 (4.8)	10.6 (2.2)	4.7 (0.7)	3.3 (0.5)	*3.1 (0.9)
Sex								
Male	6.5 (1.0)	11.7 (2.3)	2.9 (0.5)	21.0 (5.2)	10.1 (2.1)	4.2 (0.8)	2.5 (0.6)	*2.9 (1.2)
Female	5.6 (0.6)	11.9 (2.7)	4.0 (0.5)	23.0 (5.7)	11.2 (2.8)	5.0 (0.9)	3.7 (0.5)	*3.3 (1.3)
Race[2]								
White	5.1 (0.8)	10.4 (2.3)	3.3 (0.5)	23.1 (6.4)	8.8 (1.8)	4.0 (0.8)	3.2 (0.5)	*2.7 (1.0)
Male	5.3 (0.9)	10.2 (2.3)	2.7 (0.6)	*23.1 (7.1)	8.1 (1.8)	3.7 (0.9)	2.3 (0.7)	*2.5 (1.1)
Female	4.8 (0.7)	10.8 (2.6)	3.6 (0.6)	*25.2 (7.7)	9.9 (2.9)	4.3 (1.0)	3.6 (0.6)	*2.9 (1.4)
Black	11.1 (2.4)	17.6 (4.6)	5.7 (1.3)	*23.9 (7.3)	17.4 (5.2)	7.8 (2.1)	*4.8 (1.5)	*
Male	12.4 (2.9)	18.4 (4.7)	*3.7 (1.6)	*21.8 (7.9)	16.3 (5.0)	*6.7 (2.6)	*3.6 (1.8)	*
Female	10.1 (2.3)	*16.8 (5.2)	6.6 (1.6)	*27.1 (9.5)	*16.3 (6.6)	8.6 (2.4)	*5.3 (1.8)	*
Other races	4.0 (1.0)	*5.1 (1.6)	*3.2 (1.1)	*10.0 (4.9)	*4.1 (1.4)	*4.2 (2.0)	*1.6 (0.7)	*
Male	*4.2 (1.3)	*4.5 (1.9)	*3.9 (1.7)	*	*3.3 (1.1)	*	*1.3 (0.8)	*
Female	*3.7 (1.2)	*5.9 (2.6)	*2.8 (1.2)	*	*5.2 (2.4)	*	*	*
Ethnicity[3]								
Hispanic or Latino	9.9 (2.2)	13.4 (3.7)	7.4 (2.0)	*18.9 (5.8)	*12.3 (3.8)	*7.5 (2.3)	*5.8 (2.1)	*13.9 (6.9)
Male	8.3 (2.2)	13.0 (3.8)	*	*19.0 (7.3)	*11.5 (3.5)	*	*	*
Female	11.3 (2.8)	13.9 (4.1)	10.1 (2.7)	*18.7 (6.6)	*13.3 (5.2)	*10.9 (3.9)	*6.9 (2.6)	*
Not Hispanic or Latino	5.4 (0.8)	11.5 (2.5)	3.2 (0.4)	22.6 (5.5)	10.2 (2.3)	4.3 (0.7)	3.0 (0.4)	*2.3 (0.8)
Male	6.2 (1.0)	11.5 (2.5)	2.9 (0.5)	21.7 (5.9)	9.8 (2.3)	4.4 (0.8)	2.4 (0.6)	*2.9 (1.2)
Female	4.9 (0.7)	11.4 (2.8)	3.3 (0.5)	24.3 (6.8)	10.7 (2.8)	4.2 (0.8)	3.4 (0.5)	*
Region								
Northeast	8.5 (1.8)	*14.0 (4.4)	6.4 (1.4)	*29.2 (11.9)	13.7 (5.7)	7.8 (1.8)	5.2 (1.5)	*5.3 (2.5)
Midwest	6.2 (1.1)	10.3 (2.3)	4.5 (1.1)	*20.9 (6.5)	8.9 (2.2)	*4.8 (1.6)	4.8 (1.1)	*
South	*6.3 (2.0)	*13.3 (5.4)	3.0 (0.8)	*18.9 (8.4)	*12.6 (5.2)	4.7 (1.3)	2.4 (0.7)	*
West	*3.0 (1.2)	*	1.2 (0.3)	*	*	*1.8 (0.7)	1.3 (0.4)	*

* Figure does not meet standards of reliability or precision. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE over 50%.

[1]Crude estimates are shown.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race. Ethnicity was not imputed in 2001 or 2002. It is not shown for those years due to a high rate of missing data.

SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

Table 13. Rate of emergency department visits with asthma as the first-listed diagnosis per 10,000 population, by year and selected patient characteristics: United States, 2001–2009

Characteristic	2001	2002	2003	2004	2005	2006	2007	2008	2009
	Rate (standard error)[1]								
Total	59.7 (3.4)	67.9 (3.8)	61.6 (3.5)	64.0 (4.0)	61.7 (3.8)	55.3 (3.4)	59.6 (3.9)	64.5 (4.3)	69.7 (4.6)
Sex									
Male	46.7 (4.0)	65.5 (5.3)	55.0 (4.9)	62.0 (5.4)	54.7 (4.9)	47.8 (4.5)	50.5 (4.8)	60.7 (6.0)	69.5 (6.7)
Female	71.7 (5.1)	69.6 (5.1)	67.9 (4.8)	65.3 (5.0)	68.3 (5.5)	62.2 (4.7)	67.7 (5.4)	67.0 (5.3)	69.2 (5.9)
Race[2]									
White	48.2 (3.5)	46.9 (3.7)	48.9 (3.6)	43.6 (3.5)	44.4 (3.6)	39.1 (3.4)	43.7 (3.7)	44.0 (3.7)	51.0 (4.4)
Black	153.6 (14.7)	218.8 (18.2)	151.9 (13.1)	195.0 (18.7)	175.6 (17.7)	157.5 (14.6)	167.0 (16.5)	180.4 (17.7)	182.1 (19.3)
Other races	*22.2 (6.8)	23.8 (5.9)	26.6 (7.3)	39.1 (9.2)	42.1 (11.5)	45.4 (11.2)	26.2 (7.2)	55.0 (13.6)	70.5 (17.0)
Ethnicity[3]									
Hispanic or Latino	...	...	89.1 (11.8)	57.5 (7.8)	73.1 (10.8)	64.8 (9.7)	58.1 (10.2)	59.7 (9.7)	72.3 (10.2)
Not Hispanic or Latino	...	...	58.4 (3.6)	65.2 (4.4)	59.0 (4.1)	55.0 (3.8)	61.2 (4.3)	66.7 (4.6)	68.7 (5.0)
Age									
Children (0–17)	91.6 (11.2)	99.7 (13.7)	95.6 (10.3)	103.2 (14.4)	102.4 (11.3)	80.7 (9.5)	87.2 (12.9)	103.1 (13.8)	111.5 (15.3)
Adults (18 or over)	49.0 (3.5)	56.9 (4.2)	49.6 (4.3)	50.4 (5.0)	46.8 (4.1)	45.3 (4.0)	49.8 (4.6)	50.7 (5.0)	54.8 (5.4)
Age group									
0–4 years	136.6 (24.1)	159.7 (31.0)	159.1 (26.3)	168.3 (28.2)	162.7 (23.4)	115.1 (17.5)	122.1 (25.1)	145.1 (32.6)	131.5 (25.9)
5–14 years	85.4 (11.7)	79.2 (11.1)	67.0 (8.0)	82.1 (12.9)	86.3 (12.9)	64.5 (9.1)	75.8 (13.9)	101.4 (12.8)	123.0 (19.1)
15–34 years	62.8 (6.2)	74.9 (7.6)	70.3 (7.0)	73.6 (8.8)	63.1 (7.1)	57.8 (7.0)	62.2 (7.2)	63.6 (9.5)	64.9 (8.5)
35–64 years	43.6 (4.2)	53.3 (4.9)	44.8 (4.7)	42.8 (5.5)	43.8 (5.2)	48.6 (5.6)	50.7 (6.1)	43.7 (5.1)	56.5 (6.8)
65 years or over	31.2 (6.5)	32.1 (7.8)	30.8 (7.3)	27.8 (6.1)	23.1 (5.3)	17.8 (4.3)	25.0 (5.5)	40.9 (8.5)	22.8 (5.2)
Region									
Northeast	86.6 (8.2)	78.2 (6.7)	94.6 (8.5)	93.7 (9.5)	81.5 (8.3)	73.4 (8.3)	73.1 (8.3)	89.0 (12.9)	102.1 (12.2)
Midwest	66.9 (7.8)	67.2 (8.3)	47.2 (7.1)	73.8 (9.5)	64.7 (9.0)	62.7 (8.7)	58.4 (8.2)	75.0 (10.6)	85.1 (11.4)
South	54.9 (5.7)	76.9 (7.4)	62.5 (5.9)	61.5 (7.0)	56.9 (6.3)	48.3 (5.1)	65.1 (7.2)	62.1 (6.3)	63.5 (7.6)
West	37.4 (6.2)	44.8 (6.7)	47.2 (6.8)	33.7 (5.4)	50.6 (7.3)	44.4 (7.5)	42.2 (7.2)	40.2 (6.1)	40.1 (6.7)

* Figure does not meet standards of reliability or precision. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE over 50%.

... Category not applicable.

[1]Except for age-specific estimates, rates are age adjusted to the 2000 U.S. standard population.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race. Ethnicity was not imputed in 2001 or 2002. It is not shown for those years due to a high rate of missing data.

SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

Table 14. Rate of emergency department visits with asthma as the first-listed diagnosis per 100 persons with current asthma, by year and selected patient characteristics: United States, 2001–2009

Characteristic	2001	2002	2003	2004	2005	2006	2007	2008	2009
					Rate (standard error)[1]				
Total	8.2 (0.6)	9.5 (0.8)	8.8 (0.7)	8.9 (0.8)	8.0 (0.6)	7.0 (0.6)	7.7 (0.7)	8.2 (0.8)	8.4 (0.8)
Sex									
Male	7.6 (0.8)	10.8 (1.1)	9.4 (1.1)	10.0 (1.2)	8.6 (0.9)	6.9 (0.8)	7.7 (0.9)	8.5 (1.0)	9.9 (1.2)
Female	8.6 (0.7)	8.5 (0.7)	8.4 (0.7)	8.1 (0.9)	7.5 (0.7)	7.0 (0.7)	7.6 (0.9)	7.8 (0.8)	7.4 (0.8)
Race[2]									
White	6.7 (0.6)	8.7 (0.7)	7.3 (0.7)	6.3 (0.7)	5.9 (0.5)	5.0 (0.5)	5.8 (0.7)	5.9 (0.6)	6.4 (0.7)
Black	19.4 (2.5)	25.0 (3.0)	18.2 (2.3)	23.5 (3.3)	19.3 (2.6)	18.4 (2.3)	18.1 (2.6)	19.5 (2.7)	17.6 (2.7)
Other races	*3.1 (1.2)	3.7 (1.0)	*3.6 (1.3)	5.0 (1.4)	*5.5 (1.7)	*4.1 (1.4)	*3.0 (0.9)	*5.5 (2.0)	*7.6 (3.0)
Ethnicity[3]									
Hispanic or Latino	...	...	16.1 (2.6)	12.7 (2.3)	13.7 (2.5)	10.1 (1.8)	8.6 (1.7)	10.6 (2.0)	13.3 (2.4)
Not Hispanic or Latino	...	...	7.9 (0.7)	8.5 (0.8)	7.2 (0.6)	6.5 (0.6)	7.5 (0.7)	7.9 (0.7)	7.7 (0.8)
Age									
Children (0–17)	10.5 (1.3)	12.0 (1.7)	11.2 (1.3)	12.2 (1.6)	11.5 (1.3)	8.7 (1.1)	9.6 (1.5)	11.0 (1.6)	11.6 (1.7)
Adults (18 or over)	7.2 (0.5)	8.4 (0.7)	7.8 (0.7)	7.5 (0.8)	6.5 (0.6)	6.2 (0.6)	6.9 (0.7)	7.0 (0.7)	7.1 (0.7)
Age group									
0–4 years	23.9 (4.6)	26.8 (5.7)	26.8 (4.9)	30.0 (5.6)	24.1 (4.0)	19.9 (3.6)	18.0 (4.1)	23.8 (5.9)	20.9 (4.6)
5–4 years	8.6 (1.2)	8.7 (1.3)	7.1 (0.9)	9.0 (1.50	8.8 (1.4)	5.9 (0.9)	8.1 (1.6)	9.3 (1.3)	11.0 (1.8)
15–34 years	7.9 (0.6)	9.9 (1.1)	9.8 (1.1)	10.4 (1.3)	8.4 (1.0)	7.4 (1.0)	7.4 (1.0)	7.9 (1.3)	8.4 (1.2)
35–64 years	6.6 (0.7)	7.8 (0.8)	7.0 (0.8)	6.3 (0.8)	6.2 (0.8)	6.7 (0.8)	7.3 (0.9)	6.2 (0.8)	7.2 (0.9)
65 years or over	5.2 (1.1)	5.5 (1.4)	5.4 (1.3)	4.0 (0.9)	3.0 (0.7)	2.5 (0.6)	3.3 (0.8)	5.7 (1.2)	3.0 (0.7)
Region									
Northeast	10.3 (1.3)	9.3 (1.1)	11.8 (1.6)	12.4 (2.0)	9.2 (1.2)	8.1 (1.4)	6.8 (1.5)	10.7 (2.1)	11.0 (2.0)
Midwest	8.8 (1.6)	9.4 (1.7)	6.1 (1.2)	9.8 (2.1)	8.1 (1.5)	7.2 (1.4)	6.7 (1.5)	8.3 (1.7)	9.0 (1.9)
South	7.7 (1.0)	11.3 (1.6)	10.0 (1.4)	9.1 (1.5)	7.7 (1.1)	6.5 (1.1)	8.7 (1.2)	8.9 (1.2)	8.6 (1.5)
West	5.9 (0.8)	6.7 (1.2)	7.4 (1.4)	4.7 (1.0)	6.9 (1.3)	6.2 (1.3)	6.2 (1.8)	5.1 (1.1)	5.2 (1.2)

* Figure does not meet standards of reliability or precision. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE greater than 50%.

... Category not applicable.

[1]Crude rates are shown.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race. Ethnicity was not included in 2001 or 2002. It is not shown for those years due to a high rate of missing data.

SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

Table 15. Rate of emergency department visits with asthma as the first-listed diagnosis per 100 persons with current asthma, by detailed age group and selected patient characteristics: United States, average annual 2007–2009

Characteristic	Total	0–17 years	18 years or over	0–4 years	5–14 years	15–34 years	15–19 years	20–24 years	25–34 years	35–64 years	65 years or over
	Rate (standard error)[1]										
Total	8.1 (0.5)	10.7 (1.0)	7.0 (0.5)	20.8 (3.0)	9.5 (1.0)	7.9 (0.7)	6.5 (1.0)	9.1 (1.3)	8.2 (1.0)	6.9 (0.6)	4.0 (0.6)
Sex											
Male	8.7 (0.7)	11.8 (1.3)	6.7 (0.6)	22.6 (3.6)	10.4 (1.3)	6.9 (1.0)	5.7 (1.3)	7.5 (1.9)	7.7 (1.4)	7.0 (0.8)	2.6 (0.6)
Female	7.6 (0.6)	9.3 (1.1)	7.2 (0.6)	17.6 (3.3)	8.2 (1.2)	8.6 (0.9)	7.2 (1.3)	10.2 (1.7)	8.5 (1.2)	6.9 (0.6)	4.7 (0.8)
Race[2]											
White	6.1 (0.5)	9.4 (1.0)	5.0 (0.4)	19.7 (3.5)	8.7 (1.1)	5.9 (0.6)	5.0 (1.0)	6.5 (1.1)	6.1 (0.9)	4.6 (0.5)	3.2 (0.5)
Male	6.6 (0.6)	10.7 (1.4)	4.4 (0.5)	22.3 (4.6)	10.0 (1.6)	4.7 (0.9)	*3.9 (1.4)	4.5 (1.3)	5.5 (1.4)	4.2 (0.7)	2.4 (0.6)
Female	5.7 (0.5)	7.4 (1.2)	5.3 (0.5)	14.6 (3.9)	6.7 (1.3)	6.8 (0.9)	6.0 (1.4)	8.2 (1.8)	6.4 (1.0)	4.9 (0.6)	3.7 (0.7)
Black	18.3 (1.9)	15.2 (2.1)	20.9 (2.4)	24.2 (4.6)	13.1 (2.1)	19.6 (3.3)	13.9 (3.2)	23.9 (5.9)	22.6 (5.1)	21.6 (3.0)	*10.2 (3.3)
Male	19.7 (2.4)	16.3 (2.5)	24.7 (3.9)	24.2 (5.3)	14.5 (2.6)	21.0 (4.8)	14.3 (4.3)	28.2 (13.0)	*27.7 (9.8)	26.1 (5.2)	*
Female	17.3 (1.9)	13.8 (2.6)	19.1 (2.5)	24.2 (6.3)	11.4 (2.9)	18.7 (3.5)	13.6 (4.3)	21.6 (5.9)	20.7 (5.2)	19.6 (2.9)	*11.2 (4.2)
Other races	5.4 (1.4)	7.6 (2.0)	4.0 (1.6)	*17.2 (6.4)	*5.2 (1.7)	*3.9 (1.7)	*	4.1 (2.0)	*	*3.1 (1.3)	*
Male	5.0 (1.5)	*7.1 (2.3)	*	*20.1 (8.5)	*2.9 (1.2)	*	*	*	*	*	*
Female	*5.7 (1.8)	*8.4 (2.6)	*4.6 (1.9)	*	*8.3 (3.3)	*5.2 (2.6)	*	*	*	*3.1 (1.3)	*
Ethnicity[3]											
Hispanic or Latino	10.8 (1.3)	12.5 (2.1)	9.6 (1.3)	19.5 (4.9)	11.1 (2.4)	11.1 (2.0)	*7.2 (3.0)	*16.7 (5.6)	10.5 (2.6)	7.9 (1.4)	*7.8 (2.9)
Male	12.5 (2.0)	15.4 (2.9)	9.2 (2.1)	23.2 (6.1)	14.7 (3.9)	*9.6 (2.9)	*	*	*8.7 (3.8)	*7.5 (2.8)	*
Female	9.3 (1.5)	8.4 (2.2)	9.8 (1.7)	*12.9 (5.5)	6.4 (1.8)	12.2 (2.8)	*	*17.0 (6.7)	*11.7 (3.6)	8.1 (1.7)	*10.2 (4.5)
Not Hispanic or Latino	7.7 (0.5)	10.3 (1.1)	6.7 (0.5)	21.2 (3.6)	9.2 (1.0)	7.5 (0.7)	6.4 (1.0)	8.2 (1.2)	7.9 (1.0)	6.8 (0.6)	3.7 (0.5)
Male	8.1 (0.7)	11.0 (1.4)	6.4 (0.6)	22.4 (4.2)	9.5 (1.3)	6.5 (0.9)	5.6 (1.4)	6.5 (1.6)	7.5 (1.6)	6.9 (0.8)	2.5 (0.6)
Female	7.4 (0.6)	9.5 (1.2)	6.9 (0.6)	19.0 (4.0)	8.6 (1.4)	8.1 (0.9)	7.1 (1.4)	9.4 (1.7)	8.1 (1.2)	6.7 (0.6)	4.3 (0.7)
Region											
Northeast	10.2 (1.3)	12.5 (2.2)	9.2 (1.2)	*20.6 (6.3)	12.8 (2.4)	11.3 (2.2)	11.7 (2.7)	*15.1 (4.9)	9.0 (2.1)	8.1 (1.2)	*5.6 (1.7)
Midwest	8.0 (1.3)	10.9 (2.6)	6.8 (1.0)	*22.7 (8.7)	9.6 (2.1)	6.4 (1.4)	*3.7 (1.3)	5.4 (1.6)	9.4 (2.5)	7.8 (1.2)	3.2 (0.9)
South	8.7 (0.9)	11.5 (1.6)	7.4 (0.8)	20.9 (3.9)	9.5 (1.7)	8.9 (1.3)	7.5 (2.1)	11.2 (2.5)	8.3 (1.6)	7.2 (1.0)	4.1 (1.0)
West	5.5 (0.9)	7.3 (1.6)	4.9 (0.9)	17.7 (4.7)	6.4 (1.5)	5.4 (1.1)	*3.0 (1.5)	6.7 (2.0)	5.9 (1.6)	4.7 (1.0)	*3.3 (1.0)

* Figure does not meet standards of reliability or precision. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE over 50%.

[1]Crude estimates are shown.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race.

SOURCE: CDC/NCHS, National Hospital Ambulatory Medical Care Survey.

Table 16. Rate of hospital discharges with asthma as the first-listed diagnosis per 10,000 population, by year and selected patient characteristics: United States, 2001–2009

Characteristic	2001	2002	2003	2004	2005	2006	2007	2008	2009
	Rate (standard error)[1]								
Total	16.1 (1.0)	17.0 (1.0)	19.6 (1.1)	17.0 (1.0)	16.5 (0.9)	14.7 (0.9)	15.1 (1.1)	14.4 (1.0)	15.4 (1.1)
Sex									
Male	13.3 (1.1)	13.9 (1.1)	16.3 (1.3)	14.4 (1.2)	13.3 (1.0)	12.1 (1.1)	12.5 (1.1)	11.6 (1.1)	12.7 (1.2)
Female	18.4 (1.1)	19.4 (1.2)	22.7 (1.3)	18.9 (1.1)	19.0 (1.1)	16.8 (1.0)	17.4 (1.2)	17.4 (1.3)	17.7 (1.1)
Race[2]									
White	10.0 (0.7)	10.5 (0.8)	12.1 (0.8)	10.0 (0.7)	11.0 (0.7)	9.4 (0.7)	8.9 (0.8)	8.5 (0.8)	9.5 (0.8)
Black	30.5 (2.9)	35.8 (3.2)	39.5 (3.4)	35.5 (3.2)	27.0 (2.3)	29.3 (2.7)	29.1 (3.0)	30.7 (3.3)	28.5 (3.0)
Other races	13.3 (2.0)	10.0 (1.5)	17.5 (2.9)	18.8 (3.0)	13.6 (2.6)	7.0 (1.1)	10.5 (2.0)	13.2 (2.5)	16.3 (2.8)
Age									
Children (0–17)	26.2 (4.7)	26.9 (4.8)	31.2 (5.3)	27.0 (4.8)	22.4 (3.4)	21.0 (4.3)	21.3 (5.2)	18.1 (4.1)	19.1 (4.8)
Adults (18 or over)	12.5 (1.5)	13.4 (1.7)	16.0 (2.0)	13.6 (1.6)	14.6 (1.8)	12.9 (1.6)	13.2 (1.9)	13.9 (2.4)	14.6 (1.6)
Age group									
0–4 years	56.2 (10.9)	59.0 (11.0)	62.7 (11.5)	59.9 (11.1)	49.3 (8.0)	43.3 (9.7)	41.4 (9.7)	35.4 (7.9)	40.6 (11.6)
5–14 years	17.8 (3.2)	17.3 (3.1)	21.6 (3.5)	17.0 (3.1)	14.6 (2.4)	14.1 (3.0)	16.3 (4.5)	12.7 (3.3)	12.1 (2.9)
15–34 years	6.5 (0.9)	7.0 (1.2)	8.4 (1.3)	6.2 (1.0)	6.1 (1.0)	5.5 (0.8)	5.3 (0.9)	6.4 (1.2)	5.1 (0.9)
35–64 years	13.3 (1.7)	14.5 (1.8)	16.4 (1.9)	13.4 (1.7)	14.4 (1.9)	14.0 (1.8)	13.8 (2.1)	14.5 (2.6)	15.4 (1.9)
65 years or over	21.4 (2.6)	22.5 (3.2)	30.5 (4.5)	28.7 (4.5)	30.5 (4.8)	23.7 (3.4)	25.4 (4.0)	25.2 (5.0)	29.0 (4.3)
Region									
Northeast	19.3 (3.2)	20.5 (3.4)	31.0 (4.5)	23.5 (3.5)	22.0 (2.6)	22.1 (2.9)	24.7 (3.9)	18.9 (3.0)	21.0 (2.1)
Midwest	16.7 (2.1)	15.8 (2.1)	19.8 (2.4)	16.9 (2.2)	17.3 (2.2)	13.8 (1.7)	11.5 (1.4)	11.8 (1.6)	13.4 (1.6)
South	15.6 (1.3)	17.7 (1.4)	17.4 (1.3)	15.8 (1.3)	15.9 (1.3)	13.4 (1.1)	14.2 (1.2)	16.2 (1.7)	15.3 (1.7)
West	12.7 (2.2)	13.7 (2.1)	14.7 (2.2)	13.9 (2.4)	11.6 (2.1)	11.6 (2.6)	12.1 (2.7)	11.0 (2.1)	12.8 (2.6)

[1]Except for age-specific estimates, rates are age adjusted to the 2000 U.S. standard population.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned. Those with "unknown" race were excluded.

SOURCE: CDC/NCHS, National Hospital Discharge Survey.

Table 17. Rate of hospital discharges with asthma as the first-listed diagnosis per 100 persons with current asthma, by year and selected patient characteristics: United States, 2001–2009

Characteristic	2001	2002	2003	2004	2005	2006	2007	2008	2009
	Rate (standard error)[1]								
Total	2.2 (0.3)	2.4 (0.3)	2.9 (0.4)	2.4 (0.3)	2.2 (0.3)	1.9 (0.3)	2.0 (0.3)	1.9 (0.3)	2.0 (0.3)
Sex									
Male	2.2 (0.3)	2.3 (0.4)	2.8 (0.4)	2.3 (0.4)	2.1 (0.3)	1.8 (0.3)	1.9 (0.4)	1.7 (0.3)	1.9 (0.3)
Female	2.3 (0.3)	2.5 (0.3)	2.9 (0.4)	2.5 (0.3)	2.3 (0.3)	2.1 (0.3)	2.0 (0.3)	2.2 (0.4)	2.0 (0.3)
Race[2]									
White	1.5 (0.2)	1.6 (0.2)	1.8 (0.2)	1.5 (0.2)	1.5 (0.2)	1.3 (0.2)	1.3 (0.2)	1.2 (0.2)	1.3 (0.2)
Black	3.9 (0.8)	4.0 (0.8)	4.5 (0.8)	3.8 (0.8)	2.9 (0.5)	3.2 (0.6)	2.9 (0.6)	2.9 (0.7)	2.5 (0.6)
Other races	1.3 (0.3)	1.2 (0.3)	*3.0 (1.0)	3.1 (0.9)	1.9 (0.5)	0.8 (0.2)	*1.3 (0.5)	1.5 (0.4)	1.9 (0.5)
Age									
Children (0–17)	3.0 (0.6)	3.2 (0.6)	3.7 (0.6)	3.2 (0.6)	2.5 (0.4)	2.3 (0.5)	2.3 (0.6)	1.9 (0.4)	2.0 (0.5)
Adults (18 or over)	1.9 (0.2)	2.1 (0.3)	2.5 (0.3)	2.1 (0.3)	2.1 (0.3)	1.8 (0.2)	1.8 (0.3)	1.9 (0.3)	1.9 (0.2)
Age group									
0–4 years	9.7 (2.0)	9.8 (2.0)	10.5 (2.1)	10.7 (2.2)	7.3 (1.3)	7.5 (1.8)	6.1 (1.5)	5.8 (1.4)	*6.5 (2.0)
5–14 years	1.8 (0.3)	1.9 (0.4)	2.3 (0.4)	1.9 (0.4)	1.5 (0.3)	1.3 (0.3)	1.7 (0.5)	1.2 (0.3)	1.1 (0.3)
15–34 years	0.8 (0.1)	1.0 (0.2)	1.2 (0.2)	0.9 (0.2)	0.8 (0.1)	0.7 (0.1)	0.6 (0.1)	0.8 (0.2)	0.7 (0.1)
35–64 years	2.0 (0.3)	2.2 (0.3)	2.6 (0.3)	2.0 (0.3)	2.0 (0.3)	1.9 (0.3)	2.0 (0.3)	2.1 (0.4)	2.0 (0.3)
65 years or over	3.9 (0.5)	4.1 (0.6)	5.5 (0.9)	4.3 (0.7)	4.2 (0.7)	3.5 (0.6)	3.6 (0.6)	3.7 (0.8)	3.9 (0.6)
Region									
Northeast	*2.3 (0.9)	*2.5 (1.1)	*4.0 (1.5)	*3.3 (1.2)	*2.7 (0.9)	*2.6 (0.9)	*3.2 (1.3)	*2.6 (1.1)	2.5 (0.5)
Midwest	2.2 (0.6)	2.1 (0.6)	2.4 (0.6)	2.1 (0.6)	2.1 (0.5)	1.7 (0.5)	1.3 (0.4)	*1.3 (0.5)	1.4 (0.4)
South	2.2 (0.4)	2.6 (0.5)	2.7 (0.5)	2.4 (0.5)	2.3 (0.4)	1.8 (0.3)	2.0 (0.4)	2.5 (0.6)	2.2 (0.5)
West	*2.2 (0.8)	*2.3 (0.7)	*2.6 (0.9)	*2.0 (0.7)	*1.7 (0.6)	*1.8 (0.7)	*1.7 (0.7)	*1.4 (0.6)	*

* Figure does not meet standards of reliability or precision. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE over 50%.

[1]Crude rates are shown.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned. Those with "unknown" race were excluded.

SOURCE: CDC/NCHS, National Hospital Discharge Survey.

Table 18. Rate of hospital discharges with asthma as the first-listed diagnosis per 100 persons with current asthma, by detailed age group and selected patient characteristics: United States, average annual 2007–2009

Characteristic	Total	0–17 years	18 years or over	0–4 years	5–14 years	15–34 years	15–19 years	20–24 years	25–34 years	35–64 years	65 years or over
	Rate (standard error)[1]										
Total	2.0 (0.2)	2.1 (0.4)	1.9 (0.2)	5.2 (1.0)	1.6 (0.4)	0.7 (0.1)	0.6 (0.1)	0.6 (0.1)	0.9 (0.1)	2.0 (0.2)	3.7 (0.4)
Sex											
Male	1.8 (0.2)	2.2 (0.5)	1.5 (0.2)	5.3 (1.1)	1.6 (0.4)	0.6 (0.1)	0.5 (0.1)	0.5 (0.1)	0.6 (0.2)	1.8 (0.2)	2.6 (0.4)
Female	2.1 (0.2)	1.9 (0.4)	2.1 (0.2)	5.0 (1.1)	1.6 (0.3)	0.8 (0.1)	0.6 (0.1)	0.7 (0.1)	1.0 (0.1)	2.1 (0.2)	4.2 (0.5)
Race[2]											
White	1.3 (0.2)	1.4 (0.3)	1.2 (0.1)	4.2 (0.9)	1.1 (0.3)	0.4 (0.1)	0.3 (0.1)	0.4 (0.1)	0.5 (0.1)	1.3 (0.2)	2.3 (0.3)
Male	1.1 (0.2)	1.4 (0.4)	0.9 (0.1)	4.0 (0.9)	*1.1 (0.4)	0.3 (0.1)	*0.2 (0.1)	*0.2 (0.1)	0.4 (0.1)	1.1 (0.2)	1.7 (0.3)
Female	1.4 (0.2)	*	1.4 (0.2)	4.5 (1.3)	1.1 (0.2)	0.5 (0.1)	*0.4 (0.1)	0.6 (0.1)	0.6 (0.1)	1.4 (0.2)	2.6 (0.4)
Black	2.8 (0.3)	2.2 (0.4)	3.3 (0.4)	3.8 (0.8)	1.9 (0.4)	1.3 (0.2)	1.0 (0.3)	0.8 (0.2)	2.1 (0.5)	3.7 (0.5)	6.2 (1.2)
Male	2.6 (0.4)	2.2 (0.5)	3.1 (0.5)	3.8 (0.9)	1.9 (0.5)	1.4 (0.3)	*0.7 (0.3)	*1.2 (0.5)	*3.0 (1.2)	3.5 (0.6)	5.8 (1.7)
Female	2.9 (0.4)	2.1 (0.4)	3.4 (0.4)	3.9 (1.0)	1.7 (0.5)	1.3 (0.2)	*1.4 (0.4)	*0.7 (0.2)	1.7 (0.4)	3.9 (0.5)	6.3 (1.3)
Other races	1.6 (0.3)	1.3 (0.3)	1.7 (0.3)	2.4 (0.7)	*1.1 (0.3)	*0.4 (0.2)	*0.2 (0.1)	*	*0.8 (0.4)	1.7 (0.4)	5.7 (1.5)
Male	1.5 (0.3)	1.4 (0.4)	1.5 (0.4)	*2.2 (0.7)	*1.4 (0.5)	*0.4 (0.2)	*	*	*	1.7 (0.5)	*4.6 (2.3)
Female	1.6 (0.3)	*1.1 (0.3)	1.9 (0.4)	*2.8 (1.2)	*0.8 (0.4)	*	*	*	*	1.8 (0.5)	*6.6 (2.2)
Region											
Northeast	2.7 (0.3)	2.4 (0.5)	2.9 (0.4)	6.4 (1.6)	2.0 (0.4)	1.0 (0.2)	0.9 (0.2)	*1.0 (0.3)	1.1 (0.3)	2.9 (0.4)	5.1 (0.8)
Midwest	1.4 (0.2)	1.0 (0.2)	1.5 (0.2)	3.0 (0.7)	0.7 (0.2)	0.5 (0.1)	0.3 (0.1)	0.7 (0.2)	0.6 (0.1)	1.7 (0.3)	3.0 (0.6)
South	2.2 (0.4)	*2.1 (0.6)	2.3 (0.4)	4.3 (1.3)	*1.7 (0.5)	0.8 (0.2)	0.7 (0.2)	0.5 (0.1)	1.2 (0.2)	2.4 (0.4)	4.2 (0.9)
West	*1.6 (0.7)	*	1.7 (0.3)	*	*	*0.6 (0.2)	*0.4 (0.2)	*0.4 (0.2)	*0.7 (0.2)	1.2 (0.3)	*2.1 (0.7)

* Figure does not meet standards of reliability or precision. Estimates presented with an asterisk have a relative standard error (RSE) of 30%–50%. Estimates that are not shown have an RSE over 50%.

[1]Crude estimates are shown.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned. Those with "unknown" race were excluded.

SOURCE: CDC/NCHS, National Hospital Discharge Survey.

Table 19. Rate of deaths with asthma as the underlying cause per million population, by year and selected characteristics: United States, 2001–2009

Characteristic	2001	2002	2003	2004	2005	2006	2007	2008	2009
	Rate (standard error)[1]								
Total	15.0 (0.2)	14.8 (0.2)	14.0 (0.2)	12.8 (0.2)	12.8 (0.2)	11.7 (0.2)	11.0 (0.2)	10.6 (0.2)	10.5 (0.2)
Sex									
Male	11.8 (0.3)	12.3 (0.3)	11.4 (0.3)	10.4 (0.3)	9.7 (0.3)	9.4 (0.3)	9.0 (0.3)	8.3 (0.3)	8.4 (0.2)
Female	17.5 (0.3)	16.6 (0.3)	15.9 (0.3)	14.5 (0.3)	15.0 (0.3)	13.4 (0.3)	12.3 (0.3)	12.4 (0.3)	12.1 (0.3)
Race[2]									
White	12.2 (0.2)	12.1 (0.2)	11.4 (0.2)	10.4 (0.2)	10.4 (0.2)	9.4 (0.2)	8.8 (0.2)	8.5 (0.2)	8.4 (0.2)
Black	35.9 (1.1)	34.1 (1.1)	31.7 (1.1)	30.6 (1.0)	30.4 (1.0)	27.4 (1.0)	25.5 (0.9)	24.7 (0.9)	24.9 (0.8)
Other races	17.4 (0.9)	14.9 (0.8)	16.1 (0.8)	13.2 (0.7)	12.9 (0.7)	12.4 (0.7)	12.5 (0.7)	10.9 (0.6)	11.2 (0.9)
Ethnicity[3]									
Hispanic or Latino	13.9 (0.7)	13.2 (0.6)	13.0 (0.6)	11.4 (0.6)	10.1 (0.5)	9.9 (0.5)	8.3 (0.4)	9.4 (0.5)	6.0 (0.4)
Not Hispanic or Latino	15.2 (0.2)	15.0 (0.2)	14.1 (0.2)	13.0 (0.2)	13.1 (0.2)	12.0 (0.2)	11.4 (0.2)	10.9 (0.2)	10.7 (0.2)
Age									
Children (0–17)	2.6 (0.2)	3.0 (0.2)	2.7 (0.2)	2.5 (0.2)	2.3 (0.2)	2.3 (0.2)	2.5 (0.2)	2.6 (0.2)	2.8 (0.2)
Adults (18 or over)	19.2 (0.3)	18.8 (0.3)	18.0 (0.3)	16.5 (0.3)	16.7 (0.3)	15.3 (0.3)	14.4 (0.3)	14.0 (0.3)	14.0 (0.3)
Age group									
0–4 years	2.1 (0.3)	2.4 (0.4)	2.2 (0.3)	1.8 (0.3)	2.0 (0.3)	1.6 (0.3)	2.2 (0.3)	2.0 (0.3)	2.0 (0.3)
5–14 years	2.4 (0.2)	3.0 (0.3)	2.7 (0.3)	2.6 (0.3)	2.4 (0.2)	2.5 (0.2)	2.7 (0.3)	2.8 (0.3)	2.8 (0.3)
15–34 years	4.7 (0.2)	5.0 (0.3)	4.8 (0.2)	4.4 (0.2)	4.1 (0.2)	4.0 (0.2)	4.0 (0.2)	3.9 (0.2)	4.5 (0.2)
35–64 years	14.7 (0.4)	14.5 (0.4)	14.2 (0.4)	12.7 (0.3)	12.7 (0.3)	12.1 (0.3)	11.0 (0.3)	10.6 (0.3)	10.9 (0.3)
65 years or over	60.7 (1.3)	58.2 (1.3)	54.4 (1.2)	51.3 (1.2)	52.4 (1.2)	46.3 (1.1)	43.4 (1.1)	42.4 (1.0)	38.8 (1.0)
Region									
Northeast	14.4 (0.5)	13.8 (0.5)	13.8 (0.5)	12.3 (0.5)	12.2 (0.5)	10.5 (0.5)	9.8 (0.4)	10.4 (0.5)	9.9 (0.4)
Midwest	15.1 (0.5)	15.7 (0.5)	13.5 (0.5)	12.9 (0.5)	13.1 (0.4)	11.5 (0.4)	11.5 (0.4)	11.1 (0.4)	11.2 (0.4)
South	14.9 (0.4)	14.2 (0.4)	13.8 (0.4)	12.6 (0.4)	12.9 (0.3)	11.4 (0.3)	10.9 (0.3)	10.3 (0.3)	10.2 (0.3)
West	16.2 (0.5)	15.9 (0.5)	15.1 (0.4)	13.8 (0.4)	13.0 (0.4)	13.5 (0.4)	11.6 (0.4)	11.1 (0.4)	11.1 (0.4)

[1]Except for age-specific estimates, rates are age adjusted to the 2000 U.S. standard population.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race.

SOURCE: CDC/NCHS, National Vital Statistics System.

Table 20. Rate of deaths with asthma as the underlying cause per 10,000 persons with current asthma, by year and selected characteristics: United States, 2001–2009

Characteristic	2001	2002	2003	2004	2005	2006	2007	2008	2009
	Rate (standard error)[1]								
Total	2.1 (0.06)	2.1 (0.06)	2.1 (0.06)	1.9 (0.05)	1.7 (0.05)	1.6 (0.05)	1.5 (0.05)	1.5 (0.05)	1.4 (0.04)
Sex									
Male	1.7 (0.07)	1.9 (0.08)	1.8 (0.08)	1.6 (0.07)	1.4 (0.06)	1.3 (0.07)	1.3 (0.07)	1.1 (0.06)	1.2 (0.06)
Female	2.4 (0.08)	2.3 (0.08)	2.2 (0.08)	2.1 (0.07)	2.0 (0.07)	1.8 (0.07)	1.6 (0.07)	1.7 (0.07)	1.5 (0.06)
Race[2]									
White	1.9 (0.06)	2.0 (0.07)	1.9 (0.06)	1.6 (0.05)	1.6 (0.05)	1.5 (0.06)	1.4 (0.06)	1.4 (0.05)	1.2 (0.05)
Black	3.8 (0.25)	3.4 (0.21)	3.2 (0.20)	3.1 (0.20)	2.9 (0.18)	2.7 (0.18)	2.3 (0.16)	2.3 (0.16)	2.2 (0.15)
Other races	1.0 (0.11)	1.0 (0.11)	1.7 (0.22)	1.3 (0.16)	1.2 (0.15)	0.7 (0.12)	0.8 (0.13)	0.7 (0.11)	0.9 (0.12)
Ethnicity[3]									
Hispanic or Latino	1.4 (0.11)	1.7 (0.15)	1.4 (0.11)	1.3 (0.11)	0.9 (0.08)	0.9 (0.09)	0.8 (0.07)	1.0 (0.09)	1.0 (0.09)
Not Hispanic or Latino	2.2 (0.06)	2.2 (0.06)	2.1 (0.07)	1.9 (0.06)	1.8 (0.05)	1.7 (0.06)	1.6 (0.06)	1.5 (0.06)	1.4 (0.05)
Age									
Children (0–17)	0.3 (0.02)	0.4 (0.03)	0.3 (0.03)	0.3 (0.02)	0.3 (0.02)	0.2 (0.02)	0.3 (0.02)	0.3 (0.02)	0.3 (0.02)
Adults (18 or over)	2.9 (0.09)	2.9 (0.09)	2.9 (0.09)	2.5 (0.08)	2.4 (0.07)	2.1 (0.08)	2.0 (0.08)	2.0 (0.07)	1.8 (0.06)
Age group									
0–4 years	0.4 (0.06)	0.4 (0.07)	0.4 (0.06)	0.3 (0.05)	0.3 (0.05)	0.3 (0.06)	0.3 (0.05)	0.3 (0.06)	0.3 (0.06)
5–14 years	0.2 (0.03)	0.3 (0.03)	0.3 (0.03)	0.3 (0.03)	0.2 (0.03)	0.2 (0.03)	0.3 (0.03)	0.3 (0.03)	0.3 (0.03)
15–34 years	0.6 (0.04)	0.7 (0.05)	0.7 (0.05)	0.6 (0.04)	0.6 (0.04)	0.5 (0.04)	0.5 (0.04)	0.5 (0.04)	0.6 (0.04)
35–64 years	2.3 (0.10)	2.2 (0.09)	2.2 (0.10)	1.9 (0.08)	1.8 (0.08)	1.7 (0.08)	1.6 (0.08)	1.5 (0.06)	1.4 (0.07)
65 years or over	11.0 (0.71)	10.7 (0.70)	9.9 (0.66)	7.7 (0.48)	7.2 (0.42)	6.9 (0.53)	6.1 (0.42)	6.2 (0.44)	5.3 (0.39)
Region									
Northeast	1.9 (0.12)	1.8 (0.11)	1.9 (0.12)	1.9 (0.12)	1.6 (0.10)	1.3 (0.11)	1.4 (0.11)	1.5 (0.12)	1.2 (0.09)
Midwest	2.0 (0.11)	2.1 (0.13)	1.7 (0.11)	1.7 (0.09)	1.6 (0.09)	1.4 (0.11)	1.4 (0.11)	1.3 (0.09)	1.2 (0.08)
South	2.1 (0.10)	2.1 (0.11)	2.2 (0.11)	1.9 (0.09)	1.9 (0.09)	1.6 (0.08)	1.6 (0.08)	1.6 (0.09)	1.5 (0.08)
West	2.5 (0.15)	2.4 (0.14)	2.5 (0.15)	1.9 (0.11)	1.9 (0.11)	2.0 (0.14)	1.6 (0.12)	1.4 (0.11)	1.5 (0.10)

[1]Crude rates are shown.
[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.
[3]Persons of Hispanic and non-Hispanic origin may be of any race.

SOURCE: CDC/NCHS, National Vital Statistics System.

Table 21. Rate of deaths with asthma as the underlying cause per 10,000 persons with current asthma, by detailed age group and selected characteristics: United States, average annual 2007–2009

Characteristic	Total	0–17 years	18 years or over	0–14 years	5–14 years	15–34 years	15–19 years	20–24 years	25–34 years	35–64 years	65 years or over
	Rate (standard error)[1]										
Total	1.4 (0.03)	0.3 (0.01)	1.9 (0.05)	0.3 (0.03)	0.3 (0.02)	0.5 (0.02)	0.3 (0.03)	0.5 (0.04)	0.7 (0.05)	1.5 (0.05)	5.8 (0.24)
Sex											
Male	1.2 (0.04)	0.3 (0.02)	1.9 (0.07)	0.3 (0.04)	0.3 (0.02)	0.7 (0.05)	0.4 (0.04)	0.7 (0.09)	1.1 (0.11)	1.7 (0.09)	4.4 (0.32)
Female	1.6 (0.04)	0.3 (0.02)	2.0 (0.05)	0.4 (0.07)	0.2 (0.03)	0.4 (0.02)	0.3 (0.04)	0.3 (0.04)	0.5 (0.04)	1.4 (0.05)	6.6 (0.33)
Race[2]											
White	1.3 (0.03)	0.2 (0.01)	1.7 (0.04)	0.2 (0.04)	0.1 (0.01)	0.3 (0.02)	0.2 (0.03)	0.3 (0.03)	0.5 (0.04)	1.1 (0.04)	5.5 (0.25)
Male	1.0 (0.04)	0.2 (0.02)	1.5 (0.06)	0.2 (0.04)	0.1 (0.02)	0.4 (0.04)	0.2 (0.04)	0.4 (0.06)	0.7 (0.08)	1.2 (0.07)	4.0 (0.32)
Female	1.5 (0.04)	0.2 (0.02)	1.8 (0.05)	0.3 (0.07)	0.1 (0.02)	0.3 (0.02)	0.2 (0.03)	0.2 (0.03)	0.4 (0.04)	1.1 (0.05)	6.4 (0.35)
Black	2.3 (0.10)	0.6 (0.05)	3.6 (0.18)	0.6 (0.09)	0.6 (0.06)	1.5 (0.12)	0.8 (0.11)	1.5 (0.23)	2.4 (0.28)	3.8 (0.27)	8.0 (0.81)
Male	2.3 (0.14)	0.7 (0.06)	4.7 (0.41)	0.5 (0.10)	0.7 (0.09)	2.3 (0.27)	0.8 (0.16)	2.8 (0.72)	4.6 (1.08)	5.0 (0.60)	8.8 (1.73)
Female	2.3 (0.11)	0.6 (0.07)	3.1 (0.18)	0.7 (0.17)	0.5 (0.07)	1.0 (0.11)	0.7 (0.15)	0.8 (0.16)	1.4 (0.20)	3.3 (0.25)	7.7 (0.91)
Other races	0.9 (0.08)	0.1 (0.03)	1.4 (0.13)	*	*	0.3 (0.05)	*	*	0.4 (0.09)	0.9 (0.12)	7.6 (1.39)
Male	0.8 (0.10)	*	1.6 (0.21)	*	*	0.3 (0.09)	*	*	*	1.4 (0.29)	5.9 (1.56)
Female	1.0 (0.11)	*	1.4 (0.15)	*	*	*	*	*	*	0.7 (0.11)	8.8 (2.11)
Ethnicity[3]											
Hispanic or Latino	0.9 (0.05)	0.2 (0.03)	1.5 (0.09)	*	0.2 (0.03)	0.6 (0.07)	0.4 (0.08)	0.5 (0.13)	0.7 (0.11)	1.2 (0.10)	5.2 (0.69)
Male	0.9 (0.07)	0.2 (0.03)	1.7 (0.18)	*	0.2 (0.04)	0.8 (0.13)	*	*	1.1 (0.28)	1.7 (0.26)	4.0 (0.97)
Female	1.0 (0.06)	0.2 (0.04)	1.3 (0.10)	*	*	0.4 (0.07)	*	*	0.5 (0.10)	0.9 (0.10)	6.0 (0.91)
Not Hispanic or Latino	1.5 (0.03)	0.3 (0.02)	2.0 (0.05)	0.4 (0.04)	0.3 (0.02)	0.5 (0.03)	0.3 (0.03)	0.4 (0.04)	0.7 (0.05)	1.5 (0.05)	5.9 (0.26)
Male	1.3 (0.04)	0.3 (0.02)	1.9 (0.07)	0.3 (0.05)	0.3 (0.03)	0.7 (0.05)	0.4 (0.05)	0.6 (0.09)	1.1 (0.12)	1.7 (0.09)	4.4 (0.34)
Female	1.7 (0.04)	0.3 (0.01)	2.0 (0.05)	0.5 (0.05)	0.3 (0.02)	0.4 (0.02)	0.3 (0.02)	0.3 (0.02)	0.2 (0.01)	1.4 (0.04)	6.6 (0.33)
Region											
Northeast	1.4 (0.06)	0.2 (0.03)	1.9 (0.10)	*	0.3 (0.04)	0.5 (0.05)	0.2 (0.05)	0.5 (0.12)	0.8 (0.11)	1.5 (0.10)	5.4 (0.46)
Midwest	1.3 (0.06)	0.3 (0.03)	1.7 (0.08)	0.3 (0.07)	0.3 (0.04)	0.5 (0.05)	0.3 (0.05)	0.4 (0.07)	0.7 (0.10)	1.3 (0.08)	5.9 (0.51)
South	1.6 (0.06)	0.3 (0.02)	2.2 (0.08)	0.4 (0.06)	0.3 (0.03)	0.6 (0.04)	0.3 (0.05)	0.5 (0.08)	0.8 (0.08)	1.8 (0.09)	5.9 (0.43)
West	1.5 (0.07)	0.3 (0.03)	1.9 (0.10)	*	0.3 (0.04)	0.5 (0.05)	0.4 (0.08)	0.3 (0.06)	0.6 (0.08)	1.3 (0.08)	6.0 (0.53)

* Figure does not meet standards of reliability or precision. Estimates are not shown if there were fewer than 20 deaths.

[1]Crude estimates are shown.

[2]Race categories "White" and "Black" include only those with a single race. The category "Other races" includes Asian, American Indian and Alaska Native, Native Hawaiian and Other Pacific Islander, those with more than one race, and those with a race other than the groups mentioned.

[3]Persons of Hispanic and non-Hispanic origin may be of any race.

SOURCE: CDC/NCHS, National Vital Statistics System.

Appendix I. Data Sources and Limitations

This report presents data on asthma from ongoing data collection systems of the Centers for Disease Control and Prevention's (CDC) National Center for Health Statistics (NCHS). For an overview of NCHS data systems, see "Surveys and Data Collection Systems," available from http://www.cdc.gov/nchs/surveys.htm.

National Health Interview Survey

Asthma prevalence data (1980–1996, 2001–2009) and asthma attack prevalence data (1997–2009) were obtained from the National Health Interview Survey (NHIS).

Sample design

NHIS is a cross-sectional household interview survey of the U.S. civilian noninstitutionalized population. Data are collected continuously throughout the year in all 50 states and the District of Columbia. NHIS uses a multistage, clustered sample design to produce national estimates for a variety of health indicators. Information on basic health topics is collected for all household members. Interviews are conducted in the home by field staff from the U.S. Census Bureau using a computer-assisted personal interview, with telephone interviewing permitted for follow-up, if necessary. During 2001–2005, households with black and Hispanic persons were oversampled; starting in 2006, the NHIS sample design also oversampled Asian persons. From 2001 to 2009, the final response rate for each survey component, which takes household nonresponse into account, ranged from 81.6%–88.1% for the family questionnaire, 72.3%–81.3% for the sample child, and 62.6%–74.3% for the sample adult. More information can be found at http://www.cdc.gov/nchs/nhis.htm.

Item nonresponse

From 2001 through 2009, item nonresponse for each of the demographic indicators shown in this report was generally less than 1%, with the exception of poverty status, which is based on detailed family income ascertained from the family component of the questionnaire. Because the weighted percentages of persons with unknown family income were relatively high (24%–34% from 1997 through 2009), family income data were imputed using multiple imputation, with five sets of values created to allow the assessment of variability due to imputation. The ratio of family income to the applicable federal poverty thresholds were derived for families with missing incomes based on imputed income values (21). The denominators for asthma prevalence exclude persons with unknown information about asthma status (less than 0.8% for all years 1997–2009).

National Ambulatory Medical Care Survey

Physician office visit data (2001–2009) were obtained from the National Ambulatory Medical Care Survey (NAMCS).

Sample design

NAMCS is an annual probability survey of nonfederal, office-based physicians who provide direct patient care in the 50 states and the District of Columbia, excluding radiologists, anesthesiologists, and pathologists. NAMCS collects data from a national sample of over 2,000 physicians each year who provide information on nearly 30,000 visits. Because persons with multiple visits during the year may be sampled more than once, NAMCS estimates are for visits, not persons. NAMCS uses a multistage sample design procedure and sampling weights applied to each record to provide nationally representative estimates. Visits to private, nonhospital-based clinics, and health maintenance organizations (HMOs) were within the scope of the survey, but those that occurred in federally operated facilities and hospital-based outpatient departments were excluded. A sample of office-based physicians who reported that they were in direct patient care was taken from the master files of the American Medical Association and the American Osteopathic Association. Between 2001 and 2009, the overall unweighted physician response rate ranged from 58.9% to 70.4%. More information can be found at http://www.cdc.gov/nchs/ahcd.htm.

Item nonresponse

Item nonresponse rates in NAMCS were generally low (5% or less) with some exceptions. Sex was imputed for 0.8%–4.5% of records with missing values, and birth year was imputed for 1.9%–4.1% of records in survey years 2001–2008. During this period, race was imputed for 16.9%–32.9% of records. For 2003–2009, ethnicity was imputed for 19.4%–35.0% of records, but was not imputed in 2001–2002 when between 20.5% and 25.8% of records were missing ethnicity. In contrast to the imputation method used for race and ethnicity in 2005 and previous years (random assignment of a value from a patient record form with similar characteristics based on physician specialty, geographic region, and three-digit ICD–9–CM code for primary diagnosis), a hierarchical procedure was used beginning in 2006. Records missing race (or ethnicity) were initially assigned the race from a donor record after matching donor and recipient by three-digit ICD–9–CM codes for primary diagnosis and the ZIP code of the patient making the sampled visit. If no donor was found after several matching rounds of increasing geographic size (county, state, geographic region), standard imputation procedures used in 2005 and previous years were applied. If both race and ethnicity were missing, both were imputed from the same donor. Standard errors may be underestimated because single imputation was used.

National Hospital Ambulatory Medical Care Survey

Hospital outpatient visit data (2001–2009) and hospital emergency department (ED) visit data (2001–2009) were obtained from the National Hospital Ambulatory Medical Care Survey (NHAMCS).

Sample design

NHAMCS is a national probability sample survey of in-person visits made in the United States to EDs and outpatient departments (OPDs) of nonfederal, short-stay hospitals (hospitals with an average stay of fewer than 30 days) and those whose specialty is general (medical or surgical) or children's general. EDs that operate 24 hours a day are considered within the scope of the ED component; EDs that operate fewer than 24 hours are included in the OPD component of NHAMCS. The hospital sampling frame for 2006 consisted of hospitals listed in the 1991 Verispan Hospital Database (formerly known as the SMG Hospital Database), and updated using hospital data from Verispan, L.L.C. About 500 hospitals are included in the sample from which about 1,200 outpatient clinics and 400 ED departments are selected. Because persons with multiple visits during the year may be sampled more than once, estimates are for visits, not persons. NHAMCS uses a multistage probability sample and sampling weights applied to each record to produce nationally representative estimates. Between 2001 and 2009, the overall unweighted hospital response rate ranged from 88.9% to 93.9%. More information can be found at http://www.cdc.gov/nchs/ahcd.htm.

Item nonresponse

Item nonresponse rates in NHAMCS were generally low (5% or less) with some exceptions. For survey years 2001–2009, sex was missing and imputed for 0.3%–3.1% of OPD records and birth year was imputed for 0.7%–2.1% of OPD records; sex was missing and imputed for 0.4%–1.6% of ED records and birth year was imputed for 0.8%–2.7% of ED records. During this period, race was imputed for 9.4%–15.4% of OPD records and imputed for 10.4%–15.3% of ED records. Ethnicity was imputed for 11.8%–21.5% of OPD records and imputed for 15.1%–23.8% of ED records from 2003–2008. For 2001–2002, ethnicity was not imputed and was missing for 13.3%–18.6% of OPD records and for 17.0%–17.8% of ED records. In contrast to imputation methods used for race and ethnicity in 2005 and previous years (random assignment of a value from a patient record form with similar characteristics based on physician specialty, geographic region, and three-digit ICD–9–CM code for primary diagnosis), in 2006, cases missing race or ethnicity were initially assigned a donor's value after matching the donor and recipient by three-digit ICD–9–CM codes for primary diagnosis and the ZIP code of the patient making the sampled visit. If no donor was found after several matching rounds, imputation procedures based on geographic region, OPD volume by clinic type, and three-digit ICD–9–CM codes for primary diagnosis were applied. If both race and ethnicity were missing, both were imputed from the same donor. Beginning in 2007, a new method was used to impute race and ethnicity. Race and ethnicity assignments were based, where possible, on diagnosis and patient's locality (ZIP code, state, or county of residence). A hot deck approach (i.e., filling in missing values on incomplete records using values from similar but complete records of the same dataset) was employed rather than the previously used cold deck strategy (i.e., filling in missing values on incomplete records using values from similar but complete records of the dataset from the previous year), except in cases where a matching record could not be obtained from the current data. When race or ethnicity data could not be assigned using patient locality, the new method attempted to impute within the same facility wherever possible. Failing that, imputation was based on diagnosis, hospital, type of emergency service area, immediacy, and, as a last resort, on a randomly selected record.

National Hospital Discharge Survey

Hospitalization data (1980–2009) were obtained from the National Hospital Discharge Survey (NHDS).

Sample design

NHDS collects data from a sample of inpatient records acquired from a national sample of hospitals. Because persons with multiple discharges during the year may be sampled more than once, estimates are for discharges, not persons. Only hospitals with an average length of stay of fewer than 30 days for all patients, general hospitals, and children's general hospitals are included in the survey. Federal, military, and Department of Veterans Affairs hospitals, as well as hospital units of institutions, such as prison hospitals, and hospitals with fewer than six beds staffed for patient use, are excluded. NHDS collects data from a sample of approximately 270,000 inpatient records acquired from a national sample of about 500 hospitals. Between 2001 and 2009, the overall unweighted hospital response rate ranged from 86.0% to 93.9%. More information is available at http://www.cdc.gov/nchs/about/major/hdasd/nhds.htm.

Item nonresponse

Less than 1% of records were missing the sex, age, or date of birth of the patient. If the hospital record did not state either the age or sex of patient, it was imputed by assigning an age or sex value that still maintained the known distribution of these variables according to the categories of the first-listed diagnosis. In a very few cases (less than 1% of the records), the age or sex was edited because it was inconsistent with the diagnosis. For the period 2001–2009, data for race were missing for 19% to 31% of all discharges, and no attempt was made to impute for these missing values. Among asthma discharges, the percentage of records missing data for race was lower than for

all discharges, 19% to 21% for the period 2001–2009. A study on NHDS race data determined that hospitals that did not report race were likely to have a higher proportion of white discharges than hospitals that did report race, resulting in underestimation of hospital discharges for white patients (22). Consequently, the differences between racial groups are likely to be smaller than the estimates in this report indicate. For example, using crude 2009 population-based rates, when records for unknown race are excluded, black persons had a hospitalization rate 2.8 times higher compared with white persons. If all records with unknown race are assumed to be for patients of white race, the ratio of black-to-white asthma hospitalization rates decreased to 2.0. Whereas 2.8 is likely to be an overestimation of the black-white disparity, 2.0 is likely to be an underestimation given that not all records with unknown race represent white patients. Nonresponse for ethnicity is generally higher than nonresponse for race, and was not imputed, and is not reported in this analysis.

Mortality Component of the National Vital Statistics System

Death data (1980–2009) were obtained from the National Vital Statistics System (NVSS) mortality files.

Data are based on information about underlying cause of death from all death certificates filed in the 50 states and the District of Columbia, and are processed by NCHS. Mortality statistics are based on information coded by the states and provided to NCHS through the Vital Statistics Cooperative Program and from copies of original certificates received by NCHS from state registration offices. More information is available at http://www.cdc.gov/nchs/deaths.htm.

Item nonresponse

Age was not imputed for death certificates with age not recorded or unknown (between 0.01% and 0.02% of the total deaths in 2001–2009). Records with unknown age were not included in estimates by age group, but were included in overall estimates. All records coded as "other races" (between 0.03% and 0.40% of the total deaths in 2001–2009) or as unknown, not stated, or not classifiable (between 0.07% and 0.22% of the total deaths in 2001–2009) were assigned the racial designation of the previous record. All records with Hispanic origin not stated (between 0.15% and 0.33% of the total deaths in 2001–2009) were not included in estimates by Hispanic origin, but were included in the estimates by race and for overall estimates.

Appendix II. Technical Notes

In this report, data are presented for both individual years and for the most recent 3 years combined (2007–2009 or 2008–2010) for each data source. Three years of data were aggregated to provide average annual estimates for detailed demographic subgroups. Combining 3 years of data provided more stable estimates by increasing the sample size, and consequently, reducing the standard errors. Consequently, more information about detailed demographic subgroups is provided than could have been presented if using only 1 year of data.

Counts of persons, visits, deaths, and rates are provided in this report. The counts give insight into the magnitude of the disease burden on the health care system and society. The rates control for population size differences among the demographic groups and give additional insight into varying levels of the disease, asthma severity, management, and control. Rates were calculated using two methods: population-based estimates using the estimated number of persons in the U.S. population as denominators, and risk-based estimates using the estimated number of persons with current asthma from the National Health Interview Survey (NHIS) as denominators.

Population-based Rates

The type of U.S. census population estimate for each health care use and mortality data source reflects the U.S. population potentially "exposed" to each of these outcomes. Ambulatory health care visit rates (Tables 7, 9, and 13) are based on the civilian noninstitutionalized population; hospitalization rates (Table 16) are based on the civilian population; and mortality rates (Table 19) are based on the resident population. All population-based rates are age adjusted using the 2000 U.S. population distribution in 11 age categories (see "Age Adjustment" below) when the sample size allowed (23). Population-based rates are used to measure the risk of disease in the U.S. population overall and in demographic groups.

Risk-based Rates

The average annual risk-based rate tables (Tables 6, 11, 12, 15, 18, and 21) are expressed relative to the estimate of the number of people with current asthma from NHIS for the appropriate 3-year period. The risk-based trend tables (Tables 5, 8, 10, 14, 17, and 20) are expressed relative to the estimate of the number of people with current asthma for the appropriate survey year from NHIS. Risk-based rates account for prevalence differences over time and between subgroups. They provide additional insight into understanding the time trends for outcomes during a period of increasing prevalence by restricting the denominator of the rate to the population at risk for the outcome. Risk-based rates can be used to directly compare the risk of an outcome (e.g., hospitalizations or deaths) among subgroups due to factors other than a difference in current asthma prevalence. The population reference group for the prevalence estimates (civilian noninstitutionalized population) is different than the reference groups for hospital data (civilian population) and for mortality data (residential population). Therefore, the rates per 100 persons with asthma for these two outcomes may be slight overestimates. Standard errors for the asthma-specific rates, which use the number with asthma in the denominator, rather than a population constant, were calculated using formulae for ratios assuming that covariance equals zero (24).

Age Adjustment

Except for estimates for specific age groups, data shown in Figures 1, 3, 7, 8, 14, 18, 22, 26, 27, 29, and 30, and in Tables 1, 3, 5, 7, 9, 13, 16, and 19 were age adjusted using the 2000 U.S. population provided by the U.S. Census Bureau as the standard population (23). Age adjustment was used to allow comparison among various population subgroups that have different age structures. The following age groups were used for age adjustment: under age 1, 1–4, 5–14, 15–24, 25–34, 35–44, 45–54, 55–64, 65–74, 75–84, and 85 years and over. Risk-based rates are not age adjusted. Because these rates are based on the population with asthma, prevalence differences between subgroups are already taken into account. In addition, age adjusting is used to eliminate changes in the age distribution. There have been no changes in the age distribution for those with current asthma over the past 10 years.

Estimation of Counts

The estimated national counts of persons with asthma (NHIS), number of office visits for asthma to private physicians' offices (National Ambulatory Medical Care Survey), hospital outpatient departments and emergency rooms (National Hospital Ambulatory Medical Care Survey), and asthma hospitalizations (National Hospital Discharge Survey) are calculated by weighting the number of respondents or visits by the survey weights. Asthma deaths are not based on estimates but actual counts obtained from all death certificates filed in the 50 states and the District of Columbia.

Standard Error

With the exception of asthma hospitalization rates for 1980–1987 and all asthma death rates, standard errors were calculated using SUDAAN to account for each survey's complex sample design (9).

For asthma hospitalization rates for 1980–1987, the Division of Health Care Statistics makes available tables for generalized variances curves. The curves are approximations of the relative standard errors (RSEs) that are applicable to estimates of discharges by selected patient or hospital characteristics. Because exact standard errors are not provided for every size estimate, linear arithmetic interpolation was used to obtain RSE and standard

errors for estimate sizes not included in the hospitalization RSE tables (11).

For asthma deaths compiled from a complete count of death certificate data rather than from a sample, the random variation associated with mortality statistics is assumed to follow a Poisson probability distribution (10). Therefore, the standard error is associated with the number of deaths (D):

$$SE(D) = \sqrt{\text{var}(D)} = \sqrt{D}$$

where var(D) is the variance of D. The standard error associated with crude and age-specific death rates (R) assumes that the population denominator (P) is a constant and is:

$$SE(R) = \sqrt{\text{var}(D/P)} = R/\sqrt{D}$$

For risk-based asthma death rates, the standard error of the denominator (the NHIS estimate of the number of persons with current asthma) was accounted for by using the formula for ratios and assumed independent observations (covariance equals zero) (24).

Relative Standard Error

The RSE of an estimate is calculated as:

$$\text{RSE (percent)} = (S_{Est}/Est) \cdot 100$$

where S_{Est} is the standard error of the estimate and Est is the estimate. Estimates with an RSE less than or equal to 30% are considered reliable. RSEs between 30% and 50% were used to indicate unreliable estimates in the accompanying tables. Estimates with an RSE more than 50% were not reported.

Tests of Significance

Standard errors were used in significance tests. Statistical tests performed to assess the significance of differences in estimates were two-tailed with no adjustments for multiple comparisons. The test statistic used to determine statistical significance of differences between two percentages was:

$$Z = \frac{|X_a - X_b|}{\sqrt{S_a^2 + S_b^2}}$$

where X_a and X_b are the two percentages being compared, and Sa and Sb are the standard errors of those percentages. The critical value used for two-sided tests at the 0.05 level of significance was 1.96. Terms such as "higher than" and "less than" indicate statistically significant differences. Terms such as "similar" and "no difference" indicate that the statistics being compared were not significantly different. Lack of comment regarding the difference between any two statistics does not necessarily suggest that the difference was tested and found to be not significant.

Test of significance of trends was conducted using Joinpoint software to test if an apparent change over time is statistically significant, taking into account the standard error for each data point (12). The weighted least squares regression models produce an annual percentage rate change when the log of the outcome is analyzed.

Stated Comparisons

All stated comparisons used a nondirectional, two-tailed test and were significant at the $\alpha \leq 0.05$ level. Only selected comparisons among the shown demographic groups were tested.

Analysis of Trends

Significance testing for trends over time was performed with Joinpoint software from the National Cancer Institute (NCI) (12). The software fits the simplest model to trend data while allowing "joinpoints," or points marking a change in trend over the period being analyzed. The program starts with the minimum number of joinpoints (i.e., zero, which is a straight line) and tests whether more joinpoints are statistically significant and should be added to the model up to a maximum, which was specified as two for all models included in this report. Significance tests use a Monte Carlo Permutation method. More information is available on the NCI Joinpoint website at http://surveillance.cancer.gov/joinpoint/.

Joinpoint calculates the average annual percent change for each trend segment using the underlying Joinpoint regression model. Annual percent change assumes a change at a constant rate, and is calculated as the natural log of the percent of the rate of the previous year. More information can be found at http://surveillance.cancer.gov/joinpoint/aapc.html.

For some trends, the Joinpoint software identified optimal models with more than one joinpoint. A simpler model with fewer joinpoints was chosen for presentation if the difference between the two models conveyed the same overall message (i.e., the trend segment with a statistically significant annual percent change began and ended with the same years, and the annual percent change was within 5% of that identified by the more complex model).

Appendix III. Definition of Terms

Definition of General Terms

Age—Age in years is recorded for each person. Estimates are shown for age groups under age 18, 18 or over, 0–4, 5–14, 15–34 (with subcategories 15–19, 20–24, 25–34 where sample size was sufficient), 35–64, and 65 or over.

Race—The race and ethnicity categories in this report follow the 1997 U.S. Office of Management and Budget's *Revisions to the Standards for the Classification of Federal Data on Race and Ethnicity* to the extent possible, given the multiple demographic subcategories, respective sample sizes, and consistency across the multiple data sources used in this report. The 1997 standards specify five racial groups as the minimum for data collection on race in federal statistics: American Indian or Alaska Native, Asian, Black or African American, Native Hawaiian or Other Pacific Islander, and White (25). The standards also offer respondents the opportunity to select more than one of the five groups. Hispanic or Latino origin includes persons of Mexican, Puerto Rican, Cuban, Central and South American, and other or unknown Latin American or Spanish origin. Persons of Hispanic origin may be of any race. Race categories "white" and "black" are comprised of persons who indicated only a single race group. "Other races" includes Asian, American Indian and Alaskan Native, Native Hawaiian and Other Pacific Islander, persons reporting more than one race, and persons reporting their race as something other than those listed here or above. The National Vital Statistics System (NVSS) Mortality Component conformed to the 1997 standard in 21 states in 2005, and increased to 27 states and the District of Columbia in 2007. To provide uniformity and comparability of NVSS mortality data during the transition period, before all or most of the data become available in the new multiple-race format, the responses of those for whom more than one race was reported (multiple race) were bridged to a single race (10).

Ethnicity: Hispanic or Latino—Hispanic or Latino ethnicity includes persons of Mexican, Puerto Rican, Cuban, Central and South American, or Spanish origins. Persons of Hispanic or Latino origin may be of any race. In this report, the ethnicity category includes only "Hispanic or Latino" and "Not Hispanic or Latino" because subgroups of Hispanic ethnicity are not available from most data sources. The National Health Interview Survey (NHIS) ethnicity reporting, however, includes Mexican and Puerto Rican subgroups for which estimates are included in this report. The 1997 revision of Statistical Policy Directive No. 15, *Race and Ethnic Standards for Federal Statistics and Administrative Reporting,* changed the ethnicity category name from "Hispanic" to "Hispanic or Latino," but the definition of persons in that category remained the same (25). Ethnicity information is not available in hospitalization data and detailed Hispanic subgroup information is not available in ambulatory health care use or death data.

Geographic region—In the geographic classification of the U.S. population, states are grouped into the four regions used by the U.S. Census Bureau: The northeast region includes the states of Connecticut, Maine, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, and Vermont. The Midwest region includes Illinois, Indiana, Iowa, Kansas, Michigan, Minnesota, Missouri, Nebraska, North Dakota, Ohio, South Dakota, and Wisconsin. The southern region includes Alabama, Arkansas, Delaware, the District of Columbia, Florida, Georgia, Kentucky, Louisiana, Maryland, Mississippi, North Carolina, Oklahoma, South Carolina, Tennessee, Texas, Virginia, and West Virginia. The western region includes Alaska, Arizona, California, Colorado, Hawaii, Idaho, Montana, Nevada, New Mexico, Oregon, Utah, Washington, and Wyoming.

Poverty status—The average annual NHIS tables for 2008–2010 include asthma prevalence estimates by poverty status, which is not available from National Center for Health Statistics data sources on health care use or mortality. Poverty status is based on family income and family size, and is calculated using the U.S. Census Bureau's poverty thresholds for the previous calendar year. Because of the different income questions used in the 2007 NHIS and beyond, poverty ratio estimates may not be comparable with those from earlier years. For more information, see "Income and Assets Section" on page 37 of the "2009 National Health Interview Survey (NHIS) Public Use Data Release: NHIS Survey Description," available at (ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Dataset_Documentation/NHIS/2009/srvydesc.pdf).

Definition of Asthma Terms

Prevalence and attacks—There are four different prevalence measures that can be estimated from NHIS, as shown in the table below. Before 1997, data from NHIS was used to estimate asthma 12-month period prevalence. As the result of the 1997 survey redesign, 12-month asthma period prevalence has not been measured since 1996. Beginning in 1997, two new asthma prevalence measures were introduced, lifetime asthma prevalence and asthma attack prevalence. In 2001, an additional measure, current asthma prevalence, was introduced to more completely capture asthma prevalence. None of the three asthma prevalence measures collected in 1997 and afterward are directly comparable with the pre-1997 12-month period prevalence measure.

Physician office visits—Physician office visits for asthma collected from the National Ambulatory Medical Care Survey were defined as those with an *International Classification of Diseases, Ninth Revision* (ICD–9–CM) code 493 as the first-listed diagnosis.

Table. Summary of definitions for asthma items

Definitions for National Health Interview Survey (NHIS)

Estimate	NHIS question	Questionnaire
Asthma period prevalence	During the past 12 months, did anyone in the family have asthma?	Pre-1997 NHIS respiratory conditions questionnaire
Lifetime asthma prevalence	Has a doctor or other professional ever told you that (you/your child) had asthma?	1997 forward NHIS sample child and sample adult core questionnaires
Current asthma prevalence	Has a doctor or other professional ever told you that (you/your child) had asthma? AND (Do you/does your child) still have asthma?	2001 forward NHIS sample child and sample adult core questionnaires
Asthma attack prevalence	Has a doctor or other professional ever told you that (you/your child) had asthma? AND During the past 12 months, (have you/has your child) had an episode of asthma or an asthma attack?	1997 forward NHIS sample child and sample adult core questionnaires

Definitions for National Ambulatory Medical Care Survey (NAMCS)

Estimate	ICD–9–CM code	Listed position
Asthma office visit	493	First-listed diagnosis

Definitions for National Hospital Ambulatory Medical Care Survey (NHAMCS)

Estimate	ICD–9–CM code	Listed position
Asthma outpatient visit	493	First-listed diagnosis
Asthma emergency department visit	493	First-listed diagnosis

Definitions for National Discharge Survey

Estimate	ICD–9–CM code	Listed position
Hospitalization for asthma	493	First-listed diagnosis

Definitions for Mortality Component of the National Vital Statistics System

Estimate	ICD code	Listed position
Asthma death	ICD–9 code 493	1980–1998: underlying cause of death
Asthma death	ICD–10 codes J45 and J46	1999 onward: underlying cause of death

NOTE: ICD is International Classification of Diseases.

Hospital outpatient visits and emergency department visits—An outpatient or emergency department visit for asthma based on National Hospital Ambulatory Medical Care Survey data was defined as an ICD–9–CM code 493 as a first-listed diagnosis.

Hospital discharges—A hospitalization for asthma was estimated from the National Hospital Discharge Survey and defined as a discharge record with ICD–9–CM code 493 as a first-listed diagnosis.

Mortality—Asthma deaths are reported using data from the Mortality Component of the National Vital Statistics System. Under ICD–9, which was used from 1979 through 1998, asthma deaths were defined as those with an underlying cause code 493. Under ICD–10, which has been used since 1999, asthma deaths are defined as those with underlying cause codes J45 and J46. Changes in ICD affected the comparability of data coded according to the different revisions. The asthma comparability ratio for the entire population was 0.89, which indicated that 11% of the decline in asthma mortality between 1998 and 1999 was a consequence of the ICD revision (26).

List of Abbreviations

BRFSS: Behavioral Risk Factor Surveillance System

ED: Emergency department

ICD: International Classification of Diseases

NACP: National Asthma Control Program

NAEPP: National Asthma Education and Prevention Program

NAMCS: National Ambulatory Medical Care Survey

NCEH: National Center for Environmental Health

NCHS: National Center for Health Statistics

NHAMCS: National Hospital Ambulatory Medical Care Survey

NHDS: National Hospital Discharge Survey

NHIS: National Health Interview Survey

NVSS: National Vital Statistics System

OPD: Outpatient department

SLAITS: State and Local Area Integrated Telephone Survey

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Appendix E
Draft Letter on CASAC's Review of the EPA's Second Draft Policy Assessment for the Review of the Ozone NAAQS

May 19, 2014

VIA ELECTRONIC MAIL (*stallworth.holly@epa.gov*)

Dr. H. Christopher Frey
Chair, Clean Air Scientific Advisory Committee
Science Advisory Board
U.S. Environmental Protection Agency
1200 Pennsylvania Avenue NW
Washington, DC 20460

Re: Draft Letter on CASAC's Review of the EPA's *Second Draft Policy Assessment for the Review of the Ozone National Ambient Air Quality Standards*

Dear Dr. Frey:

We write regarding the draft letter on CASAC's Review of EPA's *Second Draft Policy Assessment for the Review of the Ozone National Ambient Air Quality Standards* ("NAAQS"), which was posted on May 8, 2014 ("Draft CASAC Letter"). While we agree with some points in the letter, there are several issues on which CASAC should expand and elaborate. Those points are addressed below.

The Draft CASAC Letter states that, "[t]he Second Draft PA is not clear as to how background estimates might impact the primary and secondary standards and whether these impacts may differ regionally." EPA's analysis indicates that the impact could be quite large and, therefore, background ozone levels need to be more fully considered. The lower end of the standard that CASAC is supporting – down to 60 ppb – is very close to levels of ozone that are found naturally in some regions of the country. Also, EPA's models have shown that decreasing anthropogenic sources of ozone could actually lead to *increased* ozone in some areas (because nitrogen oxides both form and destroy ozone). By controlling human sources of ozone to achieve lower standards, many parts of the country may not be able to meet current ozone standards, partially because of naturally formed ozone. In addition, in recent years, scientists have measured increasing amounts of air pollution coming to the United States from overseas. The impacts of international emissions on ozone levels in the United States, however, have not been fully considered during the current review of the ozone NAAQS. Given these issues, EPA should calculate risks that would occur with ozone exposures above background (including impacts of international emissions), and should not set standards for which some areas will be out of attainment as a result of background ozone.

The Draft CASAC Letter also states that there is scientifically sound evidence for several health effects categories, and that this evidence calls into question the adequacy of the current standard. As discussed in the comments submitted by the undersigned to CASAC on March 13, 2014, the available evidence does not support EPA's conclusions regarding long-term ozone-related respiratory effects (including mortality), short-term ozone-related all-cause mortality, or short- and long-term ozone-related cardiovascular morbidity and mortality. For all of these endpoints, there is a lack of definitive evidence supporting an effect of ozone (as opposed to other factors). Also, there is a lack of consistency and coherence within and across studies that also calls into question a causal link. Importantly, evidence of causality for these endpoints is no stronger today than it was in 2006 during the last review of the ozone NAAQS, with epidemiology studies

mostly indicating a lack of association, and other evidence (*e.g.*, animal and mechanistic studies) providing little, if any, additional support.

It is notable that the Draft CASAC Letter states the following:

> For the health-based standard, we note that the Second Draft PA outlines key uncertainties and research that needs to be addressed for future reviews of the health based standards. Specifically, we underscore the need for research to address the characterization of the exposure-response function, the identification of population thresholds, the role of co-pollutants and temperature in modifying or contributing to ozone....

These are major data gaps that undermine EPA's and CASAC's conclusion that the evidence indicates the current standard is not health protective. The data gaps also undermine CASAC's conclusion that the likelihood of ozone effects increase at lower exposures. The idea that this uncertainty essentially disappears above 60 ppb simply is not supported by the scientific evidence.

CASAC's discussion of mode of action ("MoA") in the Draft CASAC Letter seems to relate primarily to issues with how the MoA is communicated. CASAC should more carefully evaluate whether EPA's analysis of ozone's MoA was sufficient, particularly whether the discussed MoAs could occur at exposures below the current NAAQS. This MoA analysis is critically important because it could impact the presumed causal associations in the draft Policy Assessment.

Also, CASAC supports the view that ozone effects occur down to 60 ppb based on controlled human exposure studies, and concludes that any non-statistically significant findings in these studies are a result of low statistical power. However, it is unclear from the few studies that have considered exposures down to 60 ppb whether a lack of statistical significance is a result of low power or because there is no true association. In addition, even if results were statistically significant, CASAC did not discuss issues related to the relatively large uncertainty and variability associated with lung function tests or whether the study results indicated adverse effects. Lung function deficits greater than 10% can occur after some people exercise or spend time in hot or cold climates; such deficits should not be considered adverse effects on which to base a regulatory standard. Since these effects of ozone are not adverse, that should be taken into account in the current review of the ozone NAAQS.

Finally, the Policy Assessment is based on the Integrated Science Assessment, the Health Risk and Exposure Assessment (HREA), and the Welfare Risk and Exposure Assessment ("WREA"). CASAC has brought up several issues with the second draft HREA and, perhaps more importantly, EPA has indicated that it will conduct additional analyses before finalizing the HREA. Similarly, although it did not request additional analyses in the WREA, CASAC did request revisions to the interpretation placed upon the existing analyses with regard to the degree of welfare protection provided by the current standard or by a W126 standard. The public and CASAC should have an opportunity to review the additional health risk analysis and any revision to EPA's interpretation of the analyses of welfare protection. Therefore additional drafts of the HREA and the WREA – and the Policy Assessment that is based on these documents – are necessary before any of the documents can be finalized.

In closing, we maintain that there is no new significant evidence since the last review indicating that the ozone standard should be lowered below 75 ppb. Moreover, CASAC should include 75

ppb in any range that it considers.[1] We urge CASAC to consider addressing the issues discussed herein as it finalizes its letter to EPA on the *Second Draft Policy Assessment for the Review of the Ozone National Ambient Air Quality Standards.*

If you have any questions about these comments, please contact Mary Martin, Energy, Clean Air & Natural Resources Policy Counsel at the U.S. Chamber of Commerce at 202.463.5986 or by email at mmartin@uschamber.com, or Timothy Hunt, Senior Director for Air Quality Programs at the American Forest & Paper Association and American Wood Council at 202-463-2588 or by email at tim_hunt@afandpa.org or thunt@awc.org. Thank you for your consideration.

Submitted on Behalf of,

American Chemistry Council
American Forest & Paper Association
American Petroleum Institute
American Wood Council
American Iron and Steel Institute
Corn Refiners Association
Council of Industrial Boiler Owners
National Oilseed Processers Association
Portland Cement Association
Treated Wood Council
U.S. Chamber of Commerce
Utility Air Regulatory Group

[1] Also, one CASAC member commented that one cannot really distinguish risks in 1 ppb increments. However, such a small increment could have a large impact on whether a certain area could attain an ozone standard, thereby translating into significant economic impacts. CASAC should consider recommending exposure levels at which studies were conducted, even if these levels do not coincide with the 5 ppb increments that EPA considered.

Appendix F-1
March 8-Hour Ozone Averages in Ohio and Pennsylvania 2013-2014

March 8-hour ozone averages in Ohio and Pennsylvania 2013 and 2014

Concentrations of 0.060 ppm or greater

8-Hour Ozone greater than or equal to 0.060 ppm in 2013 or 2014 during March in Ohio and Penn.			
42-055-0001	Franklin	0.060	3/11/2014
42-059-0002	Greene	0.060	3/11/2014
39-047-9991	Fayette	0.060	3/30/2013
39-121-9991	Noble	0.063	3/30/2013
42-007-0002	Beaver	0.061	3/10/2013
42-045-0002	Delaware	0.060	3/31/2013
42-059-0002	Greene	0.066	3/30/2013
42-133-0011	York	0.065	3/30/2013

Appendix F-2
April Ozone Concentrations for Ohio

8-Hour Ozone Concentrations equal to or greater than 0.060 ppm in Ohio by site for April 2013								
Site								
2013	5-Apr	6-Apr	7-Apr	20-Apr	23-Apr	26-Apr	27-Apr	Days
39-003-0009							64	1
39-007-1001				60				1
39-017-9991		61						1
39-023-0001							60	1
39-025-0022	60	61						2
39-041-0002							62	1
39-047-9991						60		1
39-049-0029							62	1
39-049-0037							60	1
39-061-0006		61						1
39-061-0010	60	62						2
39-081-0017							62	1
39-083-0002							62	1
39-085-0007							61	1
39-087-0011	66					60		2
39-087-0012	64	62						2
39-089-0005						60	62	2
39-095-0024							62	1
39-095-0027							60	1
39-095-0034							60	1
39-103-0004							61	1
39-121-9991	60						63	2
39-151-0016							60	1
39-151-0022							60	1
39-155-0011					60		63	2
39-167-0004			60				60	2
39-173-0003							61	1
Days 60 or more	5	5	1	1	1	3	19	
Days 65 or more	1	0	0	0	0	0	0	
Days 71 or more	0	0	0	0	0	0	0	
Days 76 or more	0	0	0	0	0	0	0	
Maximum	66	62	60	60	60	60	64	

8-Hour Ozone Concentrations equal to or greater than 0.060 ppm in Ohio by site for April 2014								
Site								
2014	8-Apr	10-Apr	12-Apr	17-Apr	18-Apr	20-Apr	21-Apr	Days
39-003-0009			61			65	67	3
39-007-1001							73	1
39-017-0004					63	67	67	3
39-017-0018						65	68	2
39-017-9991						74	72	2
39-023-0001			60		62	63	68	4
39-023-0003			60		62	64	68	4
39-025-0022					63	66	68	3
39-027-1002						65	70	2
39-035-0034			61			60	74	3
39-035-0064						66		1
39-035-5002			61			61	75	3
39-041-0002	60				61	67	71	4
39-047-9991					64	65	72	3
39-049-0029	61		62			69	73	4
39-049-0037							70	1
39-049-0081						63	65	2
39-085-0004						63	77	2
39-057-0006					63	64	68	3
39-061-0006					62	68	69	3
39-061-0010					66	68	71	3
39-061-0040					61		64	2
39-081-0017					60	60	73	3
39-083-0002			60			63	68	3
39-085-0003						61	78	2
39-085-0007							73	1
39-087-0011			60		61	63	68	4
39-087-0012				60	61	63	64	4
39-089-0005	60		61		63	61	70	5
39-093-0019							68	1
39-095-0027						63	66	2
39-095-0034							65	1
39-097-0007					64	62	69	3
39-099-0013							70	1
39-103-0004						60	71	2
39-109-0005					63	66	65	3
39-113-0037					64	65	70	3
39-121-9991	60			61	65		74	4
39-135-1001					64	65	65	3
39-151-0016					60	62	73	3
39-151-0022							71	1
39-151-4005							68	1
39-153-0020							66	1
39-155-0009		60	63			60	78	4
39-155-0011			60		60		73	3
39-165-0007					60	64	73	3
39-167-0004	62		60		64	61	74	5
39-173-0003						61	64	2
Days 60 or more	5	1	12	2	23	36	47	
Days 65 or more	0	0	0	0	2	15	44	
Days 71 or more	0	0	0	0	0	1	20	
Days 76 or more	0	0	0	0	0	0	3	
Maximum	62	60	63	61	66	74	78	

Appendix G
Assessing Economic Impacts of a Stricter NAAQS for Ozone

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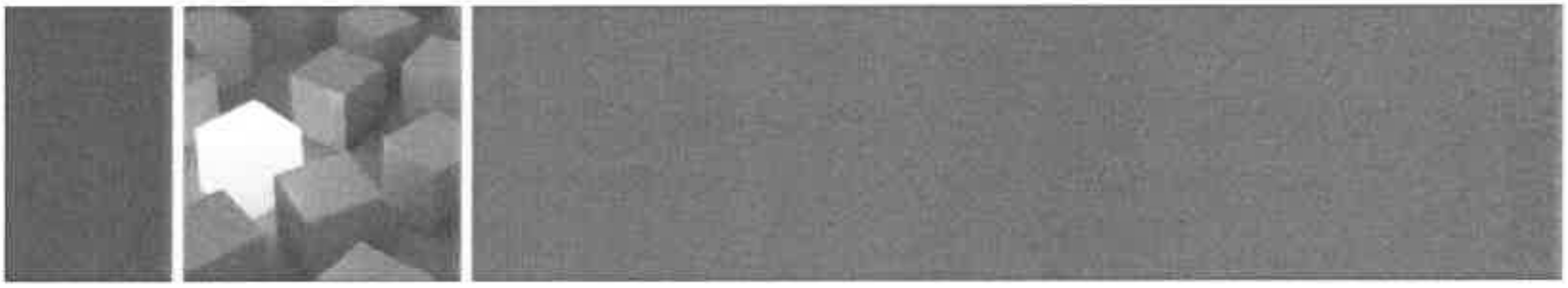
Assessing Economic Impacts of a Stricter National Ambient Air Quality Standard for Ozone



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July 2014

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EXECUTIVE SUMMARY

This report demonstrates analysis methods to assess the potential costs and impacts on the U.S. economy of a more stringent national ambient air quality standard (NAAQS) for ozone, and provides our estimates of the potential costs and economic impacts if the U.S. Environmental Protection Agency (EPA) were to set an ozone standard of 60 parts per billion (ppb). Our analysis is based on using the best available information on the emission reductions needed to attain a 60 ppb standard and the costs of those reductions; because that information is limited, we refer to our results as potential costs and economic impacts. Employing our integrated energy-economic model (N_{ew}ERA), we estimate that the potential emissions control costs would reduce U.S. Gross Domestic Product (GDP) by $270 billion per year on average over the period from 2017 through 2040 and by more than $3 trillion over that period in present value terms.[1] The potential labor market impacts represent an average annual loss of 2.9 million job-equivalents.[2]

A tighter ozone standard may also result in barriers to new energy production activity in areas that become in nonattainment. We therefore also consider a sensitivity case that includes constraints on new natural gas production in the U.S., leading to even greater estimated impacts in terms of energy costs for consumers and losses in economic output. In this sensitivity case, we estimate a GDP reduction of $360 billion on average and more than $4 trillion over the period from 2017 through 2040 in present value terms, and a projected average annual loss of 4.3 million job-equivalents.

These large potential impacts and the limited information now available to estimate them provide compelling arguments that EPA needs to provide more complete data and analysis as part of its forthcoming proposal to revise the ozone NAAQS, so that there can be better understanding of the economic impacts of the range of alternative ozone NAAQS levels. In particular, EPA needs to make a concerted effort to specify the full set of controls needed to achieve attainment of various ozone standards. A concerted effort is needed because currently EPA's "known" controls represent only one-third of the estimated reductions needed to achieve a 60 ppb standard, with the remaining two-thirds consisting of unspecified ("unknown") controls. EPA also needs to develop specific estimates of costs per ton for *all* of the needed controls, including a reliable means of extrapolating costs where cost information is not available. Finally, EPA needs to perform economy-wide modeling that accounts for both the emissions reduction costs

[1] All dollar values in this report are in 2013 dollars unless otherwise noted. The present value reflects impacts from 2017 through 2040, as of 2014 discounted at a 5% real discount rate; this discount rate falls in the 3% to 7% range recommended in EPA's *Guidelines for Preparing Economic Analyses* (2010a, p. 6-19), and it is consistent with the discount rate used in the N_{ew}ERA model.

[2] "Job-equivalents" is defined as total labor income change divided by the average annual income per job. This measure does not represent a projection of numbers of workers that may need to change jobs and/or be unemployed, as some or all of the loss could be spread across workers who remain employed, thereby impacting many more that 2.9 million workers, but with lesser impacts per worker.

that the various states would face and the potential barriers to economic development due to other regulatory consequences of being designated as a nonattainment area.

Study Objectives

Our study had two principal objectives:

1. Assess the costs and economic impacts of a 60 ppb ozone standard using the best available information from EPA and other sources; and
2. Develop recommendations for additional and updated information and analyses EPA should provide in its regulatory impact analysis (RIA) of a proposed rule, so that such assessments can be more fully evidence-based.

The first objective was predicated on the large potential significance to the U.S. economy of a more stringent ozone standard as indicated by EPA's own prior partial estimate (excluding costs in California) that the annualized costs would be $90 billion per year in 2006 dollars ($102 billion in 2013 dollars) to achieve a 60 ppb standard using one of EPA's calculation methodologies (EPA 2010b, p. S2-19).[3] Unlike regulations that target specific sectors, an ozone standard would directly affect virtually every sector of the economy, because ozone precursors (oxides of nitrogen, or NO_X, and many types of volatile organic compounds, or VOCs) are emitted by a wide range of stationary, mobile, and area sources. Moreover, a tightened standard might result in other effects, notably potential constraints on domestic natural gas and crude oil development activity if nonattainment regions introduce permitting barriers or require emissions offsets to develop new wells and processing facilities.

The second objective of this study relates to EPA's process of updating its analysis as it prepares its RIA. Our analysis reveals major gaps in information on compliance technologies and their costs and in other important information. Our research thus puts us in a position to recommend information that EPA should develop and make available in order to provide comprehensive and reliable assessments of the economic impacts of a more stringent ozone standard.

Background on the Ozone NAAQS and Its Implementation

Under the Clean Air Act, EPA is instructed to select a primary NAAQS that protects the nation's public health with an "adequate margin of safety" (Section 109(b)(1)). In March 2008, EPA lowered the primary 8-hour ozone NAAQS from 80 parts per billion (ppb) to 75 ppb.[4] EPA is

[3] Additional discussion of EPA's previous total cost estimates and differences from our cost estimates appears in Appendix C. The total estimated annualized cost of $90 billion is based on EPA's hybrid cost approach with the middle slope parameter.

[4] Due to rounding conventions, areas could comply with the ozone standard of 80 ppb with ozone levels as high as 84 ppb. The ozone primary standard is based upon the annual fourth-highest daily maximum 8-hour concentration, averaged over three years. There is also a secondary standard that presently is the same as the primary standard. We have not assigned any costs for a potentially-tighter secondary standard, should it be more stringent than the primary standard in some locations.

again reviewing the ozone NAAQS, including developing various assessment materials and obtaining advice from the Clean Air Scientific Advisory Committee (CASAC). The new NAAQS rule is expected to be proposed near the end of this year and promulgated in late 2015. EPA has stated its intention to consider tightening the standard to as low as 60 ppb in its most recent draft policy assessment for the ozone NAAQS, and CASAC has endorsed EPA considering a NAAQS in the 60 to 70 ppb range. Our study evaluates a new ozone standard of 60 ppb, one alternative standard that is likely to be included when EPA issues its proposed rule. It is intended to illustrate the types of data and analysis that EPA should undertake and present for each alternative standard that is included in its proposed rule.

After a NAAQS has been promulgated, states must review data from their ambient monitoring networks and identify areas that are not attaining the new NAAQS (called "nonattainment areas"). States must then develop state implementation plans (SIPs) that identify what sources of emissions will be reduced, and when, to achieve attainment on the regulatory schedule. For ozone, attainment will require reductions in both NO_X and many types of VOCs. In most of the U.S., NO_X reductions are presently more effective for reducing ozone formation than VOC reductions. Estimates of the cost of reducing emissions, such as EPA presented in its previous ozone NAAQS rulemakings, thus focus mostly on controls of NO_X. In the absence of any new information to the contrary, our cost analysis is also based mostly on NO_X emissions reduction needs – including what they will cost, and what sectors will pay for them.

Finally, being in nonattainment of a NAAQS triggers more regulatory burdens than just reducing emissions to achieve attainment. A number of regulatory programs are also imposed on nonattainment areas. Significant among these is a requirement that any economic entity that wishes to obtain a permit to establish a new facility that will emit the pollutant(s) of concern in a nonattainment area must first find an offsetting reduction of those same emissions from another facility that is exiting the area, or has voluntarily reduced its own emissions below its permitted level. Markets for these "offsets" often develop, but offsets can be exceedingly costly or difficult to find if there are few existing emitting facilities in the area to create a supply. A tightened ozone standard has the potential to cause nonattainment areas to expand into relatively rural areas, where there are few or no existing manufacturing facilities to generate a supply of offsets. If nonattainment expands into rural areas that are active in U.S. oil and gas extraction, a shortage of offsets may translate into a significant barrier to obtaining permits for the new wells and pipelines needed to expand (or even maintain) our domestic oil and gas production levels. Our analysis also considers the potential implications of this often-ignored aspect of nonattainment status.

Costs of Emissions Controls to Achieve Attainment

Methodology for Estimating Compliance Costs

Although EPA's review of the NAAQS for ozone is under way, EPA has not released any new ozone compliance cost estimates since its 2008-2010 analyses.[5] EPA has issued some updated information on projected baseline emissions, and there is updated monitored ozone concentration data that helps indicate the areas and states most likely to be designated in nonattainment with a 60 ppb standard. The updated information allows us to develop estimates of emissions reductions that might be required for these states to come into attainment. The cost information available from EPA, however, currently is very limited, and we explain ways that EPA needs to improve it. The data development and analysis needs that we highlight in this report will be the same whether EPA chooses to propose a standard of 60 ppb, as we analyze here, or some other level.

Our compliance cost estimates are based upon a synthesis of four major sources of information: (i) the most recent EPA information on projected 2018 baseline VOC and NO_X emissions (EPA 2014a) supplemented by baseline emission projections for electric generating units (EGUs) from N_{ew}ERA; (ii) our assessments (based upon earlier EPA analyses) of emission reductions that would be required for all regions of the United States to come into attainment; (iii) cost and emission reduction information that EPA has developed for what it refers to as "known" controls; and (iv) our estimates of the emission reductions and potential costs per ton of what EPA refers to as the "unknown" controls necessary to achieve attainment in each affected state.[6] The report and appendices provide details on our methodology. Although this report describes results for the United States as a whole and disaggregated to 11 regions,[7] the inputs and the results are built up using detailed state-specific and sector-specific cost information. The costs and impacts of a more stringent ozone standard differ substantially among states.

EPA's 2008-2010 ozone analyses identified many specific NO_X emissions controls that could be adopted by existing NO_X emissions sources in and around projected nonattainment areas. As noted, those "known" control options did not provide sufficient emissions reductions to attain a 60 ppb standard in most of the projected nonattainment areas. Indeed, the bulk of the estimated compliance costs for a 60 ppb standard in EPA's prior ozone analyses were based on extrapolations from "known" control costs to the costs of unspecified ("unknown") controls on

[5] "EPA's 2008-2010 ozone analyses" refers to information in EPA's 2008 regulatory impact analysis (RIA) for the ozone NAAQS, including information on baseline future conditions and ozone standards of 80 (effectively 84) and 75 ppb (EPA 2008); EPA's 2010 supplemental regulatory impact analysis, including information on an ozone standard of 60 ppb (EPA 2010b); and data files in Docket No. EPA-HQ-OAR-2007-0225.

[6] EPA's 2008-2010 ozone analyses refer to various types of controls. We refer to the "known" costs as those EPA refers to as "known" and "supplemental." The "unknown" controls are not specified by EPA.

[7] "U.S." results are, formally, only for the lower 48 states, and exclude Alaska and Hawaii, as well as Washington DC. We refer to the lower 48 states as "U.S." hereafter.

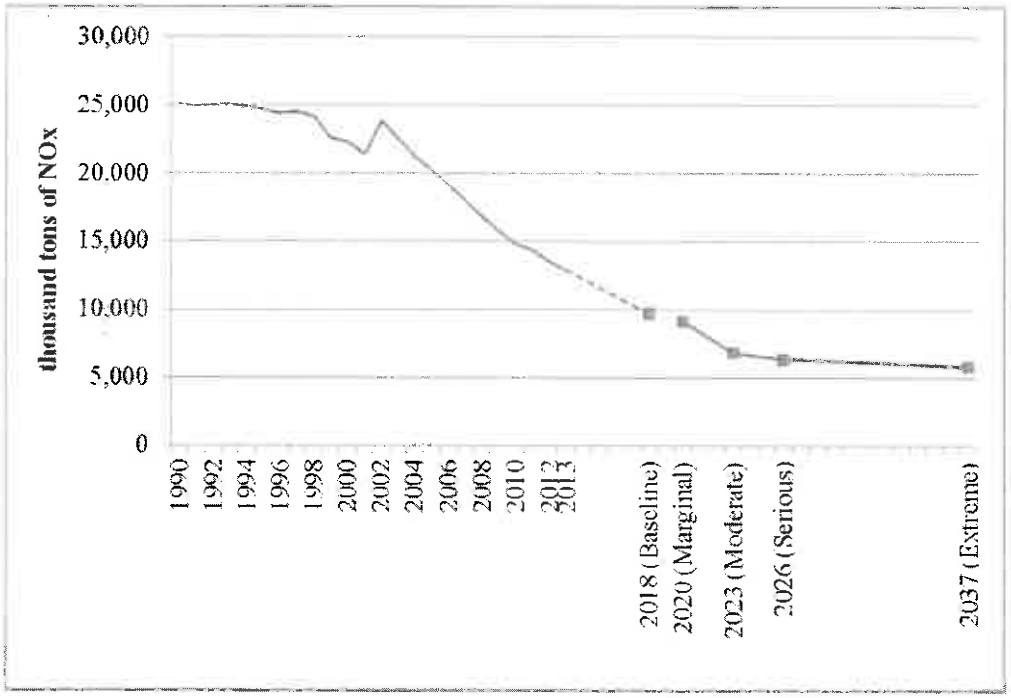
the many other, mostly dispersed, sources of NO_X in the U.S. These extrapolations relied on information on the cost per ton of "known" controls—rather than potential costs for "unknown" controls—and on largely arbitrary assumptions about the maximum cost per ton that any area would incur. Extrapolations of any sort are fraught with a high degree of uncertainty. Such uncertainty is a particular concern when the resulting compliance cost estimates are very large, as is the case for a tightened NAAQS.

In this study, we demonstrate an evidence-based approach for estimating costs of the "unknown" controls that provides one template for developing reliable estimates of total compliance costs. To do so, we first assume that the "known" control measures identified by EPA in its 2008-2010 ozone analyses have not yet been adopted, and thus will still be part of the attainment effort to meet a tightened ozone NAAQS. We then identify the types of sources and activities that account for the remaining emissions of NO_X, and the extent to which they also will need to be reduced in order to attain a 60 ppb NAAQS. By directly characterizing what sources those "unknown" controls must come from, one can develop a more informed (albeit still uncertain) procedure for estimating the total costs of attainment.

As Figure S-1 illustrates, national NO_X emissions have already been reduced substantially, from about 25.2 million tons in 1990 to 12.9 million tons in 2013 (EPA 2014b). EPA presently is projecting that U.S. NO_X emissions will be further reduced to 9.7 million tons by 2018 (supplemented with EGU baseline emission projections from N_{ew}ERA) due to existing rules and regulations, some of which have not yet been fully implemented and will carry with them additional compliance costs on top of any compliance costs estimated in this study (EPA 2014a).[8] Economic activity (as measured by real GDP) in 2018 is projected to be more than double the level in 1990 (CEA 2014, Table B-3 and OMB 2013, Table 2), suggesting that U.S. NO_X sources will have been controlled by roughly 80% by 2018, even before the additional controls needed to attain a tighter ozone NAAQS.

[8] These are national totals, but the reductions to get down to 9.7 million tons will have to occur primarily in states with nonattainment areas. As we will explain later, we estimate that 40 states will have at least some nonattainment for a new ozone NAAQS of 60 ppb. Among these 40 states, projected 2018 NO_X emissions are 8.9 million tons, and NO_X emissions need to be reduced to 5.0 million tons for attainment.

Figure S-1: U.S. NO_X Emissions to Attain 60 ppb NAAQS Compared to Historical NO_X



Notes: Blue solid line: estimated historical emissions; blue dotted line: projected further declines through 2018; Red line: emissions to attain 60 ppb on attainment schedule.
The slight increase in U.S. NO_X emissions from 2001 to 2002 primarily reflects changes in EPA's emission modeling methodology for onroad and nonroad sources (switching from MOBILE6 to the National Mobile Inventory Model and MOVES)
Source: NERA calculations as explained in text

Based upon the 2008-2010 EPA review, total U.S. NO_X emissions would have to be reduced to about 5.8 million tons to meet a 60 ppb standard throughout the nation. This reduction appears as the red line above in Figure S-1, which also shows our projection of the timing of those reductions, based on our estimates of the likely severity classifications of the different states. Despite the extensive controls already expected to have occurred through 2018, we estimate that another 3.9 million tons (in aggregate) would need to be eliminated across the 40 states that our analysis indicates would not attain a 60 ppb standard under the 2018 baseline emissions, in order for those states to come into attainment. This is equivalent to another 45% reduction from those states' 2018 NO_X emissions, and it implies about 90% total reduction from all sizes and types of NO_X-emitting sources from the relatively uncontrolled emissions rates in 1990 (after adjusting for growth).

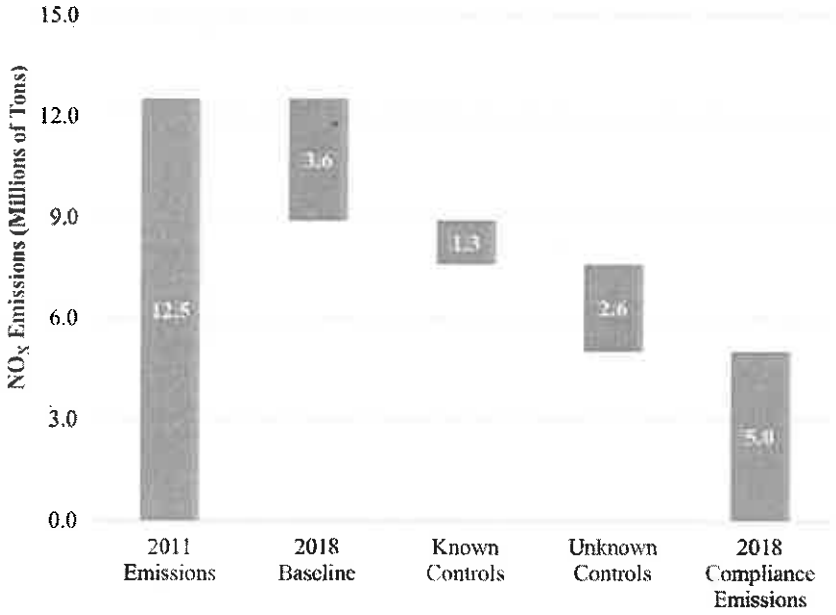
The EPA 2008-2010 analyses also imply that the EPA "known" control measures would reduce about 1.3 million tons of NO_X across those 40 non-attaining states. The remaining 2.6 million tons (two-thirds of the total necessary emission reduction) would need to come from the "unknown" controls that EPA was unable to identify in its 2008-2010 analyses. The waterfall chart of Figure S-2 summarizes the relative magnitudes of the three types of emissions

reductions our analysis projects the 40 non-attaining states would rely on to get from their recent (2011) NO_X emissions to the estimated levels needed for attainment with a 60 ppb standard. In our analysis, we treat the first reduction block – which is reductions due to changes in activity and other non-ozone regulations presently being implemented – as costless (although we include the costs of controls that have not been implemented). While regulations now being implemented clearly have a cost, we do not attribute those costs to a tightened ozone NAAQS. For EPA's list of known controls, we use EPA's earlier cost estimates. For the block of controls that EPA called "unknown," we use additional data and analysis to develop our own cost estimate.

Answers to three key questions about these "unknown" controls will determine the overall costs (and, indeed the feasibility) of a 60 ppb ozone standard.

- What categories of emission sources would be potentially available to achieve these additional 2.6 million tons of "unknown" NO_X reductions?
- What types of control strategies would likely be used for these "unknown" NO_X emission reductions?
- What would be the costs of these "unknown" controls?

Figure S-2: NO_X Emissions and Categories of NO_X Reductions to Attain 60 ppb NAAQS (for 40 Non-Attaining States Only)



Note: Emissions and reductions include only states requiring emission reductions for compliance with a new ozone NAAQS of 60 ppb in this analysis.
Source: NERA calculations as explained in text

To address the first two of the above questions, we look at EPA's estimates of the categories of 2018 baseline emissions disaggregated down to the five categories that EPA identifies in its emissions inventories, and we allocate the reductions from EPA's list of known controls to these categories.[9] This information gives insights on what types of control strategies might be available to obtain further "unknown" reductions.

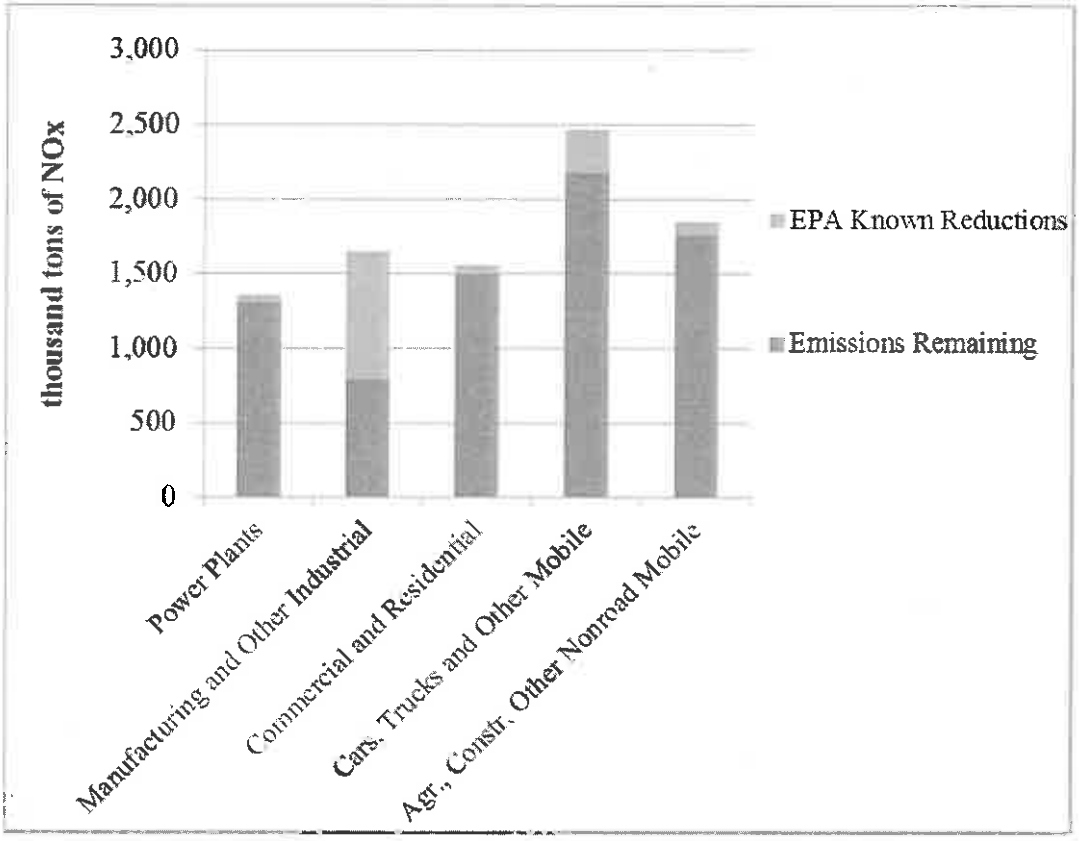
Figure S-3 shows the 2018 baseline emissions in states that will need to reduce NO_X emissions to meet a 60 ppb standard and the portion that would be eliminated by EPA's "known" controls for the five emission categories. (The total 2018 baseline emission for these states across all five categories is 8.9 million tons.) This information shows that most of the emissions that remain after EPA's "known" controls are from electricity generating units (EGUs) and the three types of non-point sources, while large industrial and manufacturing point sources are substantially controlled.

Detailed information on the list of known controls (described in the main report) indicates that the "known" controls seem to exhaust the options for retrofitting existing equipment with technology controls (e.g., installation of low-NO_X combustion devices and NO_X-destroying post-combustion devices). This explains why most of the known controls' effects are concentrated on the industrial and manufacturing emitters that comprise the "point source" category.[10] This evidence suggests that the bulk of the 2.6 million tons of "unknown" NO_X reductions will have to come various forms of capital stock replacement rather than further technology retrofits. While these replacements will likely include retirements of large coal-fired electricity generators, it also will likely become necessary to scrap and replace a wide array of very small sources, such as personal vehicles, individual pieces of construction equipment, and agricultural and landscaping equipment.

[9] The categories include two types of "point sources," which are large non-moving emitting equipment such as industrial boilers and electricity generating units (EGUs). The other three categories are "non-point sources," which means they are many small, diffuse sources. Of these "area sources" are non-moving equipment that are too individually small to be regulated as point sources are. Examples include commercial and residential water and space heaters as well as compressors along oil and natural gas pipelines. "Mobile sources" are small, diffuse and can be moved from place to place. Onroad mobile sources include cars and all sizes of trucks. Nonroad mobile sources include agricultural and construction equipment as well as transportation such as locomotives, airplanes, and boats.

[10] EPA's "known" controls for electric generation unit (EGU) sources (which are mostly from additional retrofits of selective catalytic reduction, SCR) have very little effect on EGU 2018 emissions. This is because almost all of the EGU point sources have already been retrofitted with NO_X controls in states projected to have nonattainment. Retirements rather than further retrofitting will be necessary to further reduce EGU emissions in these states and EPA did not consider retirements of equipment as a known control.

Figure S-3: EPA Known NO_X Reductions from 2008-2010 Analysis and Remaining Emissions by General Categories of Emissions Sources in the 40 Non-Attaining States



Source: NERA calculations as explained in text

To show how EPA can develop a more informed estimate of the costs of these remaining necessary types of NO_X reductions, we considered the costs of reducing emissions from two of the most significant categories of remaining NO_X emissions.

- *Retirement of coal-fired power plants.* If coal units are retired in states with large remaining NO_X reductions needs, and their generation is replaced by a cost-effective combination of natural gas and non-emitting generation, we estimate that an additional emissions reduction of about 1 million tons could be obtained. Our analyses indicate that these tons of reduction will cost an average of approximately $31,000/ton, but with costs ranging up to about $180,000/ton among the states. We replace the "known" power plant controls (retrofits) used in EPA's 2008-2010 analyses with these potential retirement controls in our analysis.

- *Scrapping of cars and light-duty trucks.* Cars and trucks will be much lower-emitting in 2018 than the fleet of vehicles on the road today, but in aggregate they account for a

further potential reduction of 1 million tons, assuming every 2018 vehicle were to be scrapped in 2018 and replaced by either an electric vehicle (powered by natural gas generation) or a Tier 3 vehicle.[11] Using a model framework developed by an MIT researcher to evaluate an existing vehicle scrapping program (Knittel 2009), we estimate the marginal cost per ton of reducing light-duty vehicle NO_X emissions by 10% through the early replacement of the highest-emitting cars and trucks would be in the range of $100,000/ton, a figure that escalates to about $500,000/ton to achieve about a 50% reduction. Scrapping newer, lower-emitting cars would cost more and generate fewer reductions per vehicle, so the incremental cost per ton rises as increasing percentages of the vehicle fleet are scrapped.

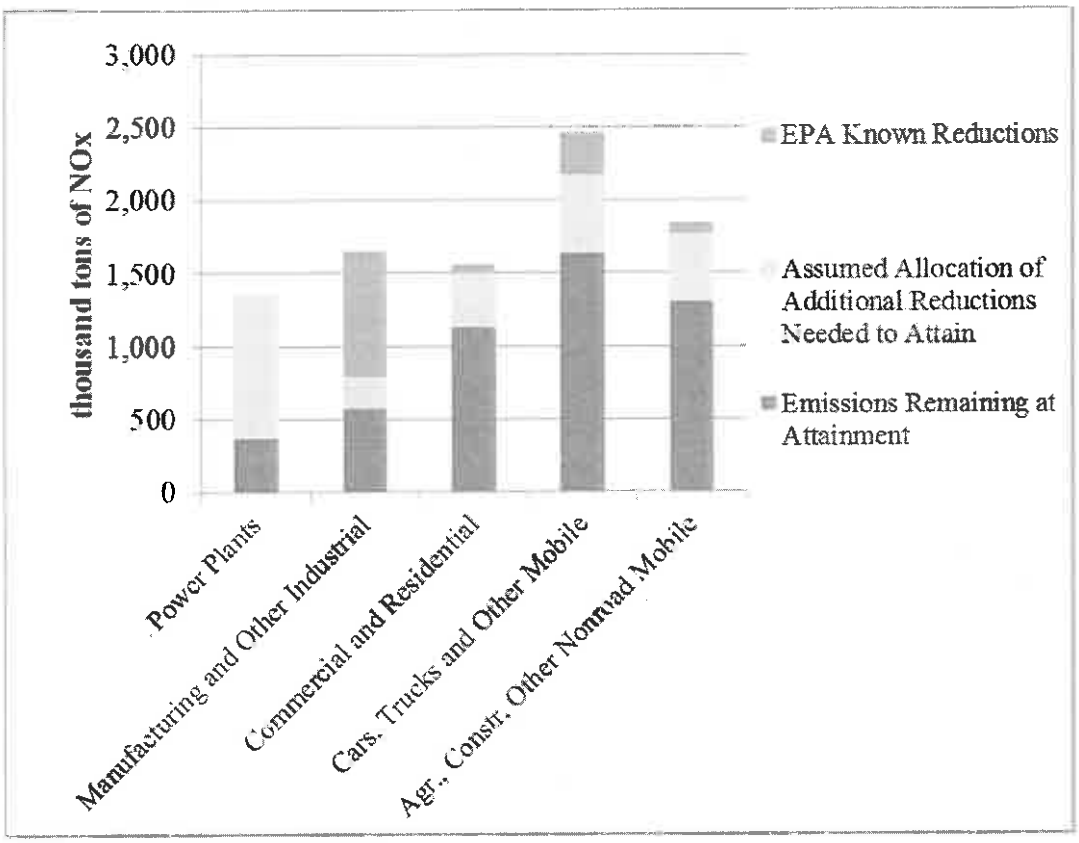
Replacing coal-fired EGUs would reduce NO_X emissions by about 1 million tons. Replacing *all* 2018 cars and light-duty vehicles would provide another 1 million tons of reduction. But other types of equipment certainly would become cost-effective to replace before one would go so far as to scrap all cars and light duty vehicles. We assume that the marginal cost-per-ton for these other sources rise similarly to those we estimated for early turnover of different vintages of cars and light-duty trucks, as one indication of the potential costs that states would incur to control the other non-point sources.

Figure S-4 shows the resulting mix of reductions assumed in our estimates of the compliance costs needed to achieve a 60 ppb ozone standard. The dark green shows EPA's "known" controls and the light green shows NERA's evidence-based assumptions regarding where "unknown" controls will likely come from. The remaining sum (shown in the blue bars) is now 5.0 million tons—the aggregate limit to achieve attainment for the states projected to be in nonattainment under baseline 2018 emissions levels. NERA's estimates assume deep cuts in the EGU sector, where emissions are concentrated in a few sources and costs per ton are thus lower than for the many smaller sources among the non-point source categories (i.e., area, onroad mobile and nonroad mobile).[12] NERA's assumptions on "unknown" controls outside of the EGU sector involve much smaller incremental percentage reductions than from EGUs; but because these will require programs such as scrapping vehicles and other small sources, they are expected to come at a substantially higher cost per ton than the EGU controls—even though we assume that the scrapping programs only target the oldest, highest-emitting of each type of NO_X-emitting equipment.

[11] The reduction is less than 1 million if one considers only vehicles in areas that contribute to nonattainment.

[12] As discussed below, EPA (2014d, pp. ES-6 and ES-7) estimates that its recently proposed power sector CO_2 rule would reduce annual NO_X emissions by approximately 300,000 to 400,000 tons (depending on regulatory option, state or regional compliance approach, and measurement year). Our modeling of potential changes to coal-fired power plants for compliance with a new ozone NAAQS of 60 ppb would lead to a significantly larger NO_X reduction (as shown in Appendix C). Thus, the proposed power sector CO_2 rule would not change our conclusion that a new 60 ppb ozone NAAQS would have significant impacts on the power sector (and other sectors of the economy).

Figure S-4: NERA Analysis's Allocation of Additional Reductions Necessary to Attain a 60 ppb NAAQS to Categories of Emissions Sources in the 40 Non-Attaining States



Source: NERA calculations as explained in text

On June 2, 2014, EPA released a proposed rule to limit CO_2 emissions from the power sector. Implementation of such a rule will almost certainly result in some amount of NO_X reduction (primarily because of a reduction in coal-fired generation needed to reduce state CO_2 emissions rates). EPA (2014d, pp. ES-6 and ES-7) estimates that the proposed power sector CO_2 rule would reduce annual NO_X emissions by approximately 300,000 to 400,000 tons (depending on regulatory option, state or regional compliance approach, and measurement year). Some of those NO_X reductions may overlap with NO_X reductions in our ozone cost analysis, and to the extent that this would occur, some of the cost estimates will be shared with the cost of the proposed CO_2 rule for the power sector. We have considered this issue and find that even if all of those overlapping costs were to be removed from our analysis, the costs and economic impacts presented in this report would not change in any meaningful degree because NO_X reductions from the power sector are estimated to be among the lowest cost-per-ton of the NO_X reductions in our ozone attainment scenario. In fact, even if all of the approximately 100 GW reduction in electric sector coal capacity (discussed in subsequent sections of this report) were treated as costless in our ozone analysis, our estimated GDP impact would only be reduced by

about 8%, or about one-twelfth of the value we estimate. Moreover, our review of EPA's RIA for the power plant CO_2 rule indicates that the overlap of power sector costs is much less than 100%, as the proposed power sector CO_2 rule would lead to NO_X reductions in some areas that would not require reductions for compliance with a new 60 ppb ozone standard. Thus, uncertainty on how to attribute shared costs between the two regulations does not affect the major conclusions of this report regarding the costs and economic impacts of a tightened ozone NAAQS.

Estimates of Potential Compliance Costs

We estimate that the potential costs of achieving a 60 ppb ozone standard would have a present value of $2.2 trillion as of 2014 (based upon costs incurred from 2017 through 2040), as reported in Figure S-5. As a rough point of comparison, EPA's annualized cost estimate implies a present value of about $0.9 trillion.[13] The primary difference in our methodologies is the extrapolation method used to estimate the cost of "unknown" controls that were not identified in EPA's 2008-2010 analyses; we attempted to understand the kinds of controls that would be required after "known" controls and based our method on the estimated costs of one such control (vehicle scrappage), whereas EPA relied on an arbitrary extension from "known" control costs. As discussed in the report, our cost estimate is still subject to substantial uncertainty. We also note that our evidence-based approach could be extended to other types of equipment that, in aggregate, make substantial contributions to non-point source NO_X emissions. Given the importance of these additional controls to the compliance cost estimate, we conclude that it would be important for EPA to develop information before it releases its proposal to revise the ozone NAAQS.

Figure S-5: Potential U.S. Compliance Spending Costs for 60 ppb Ozone Standard

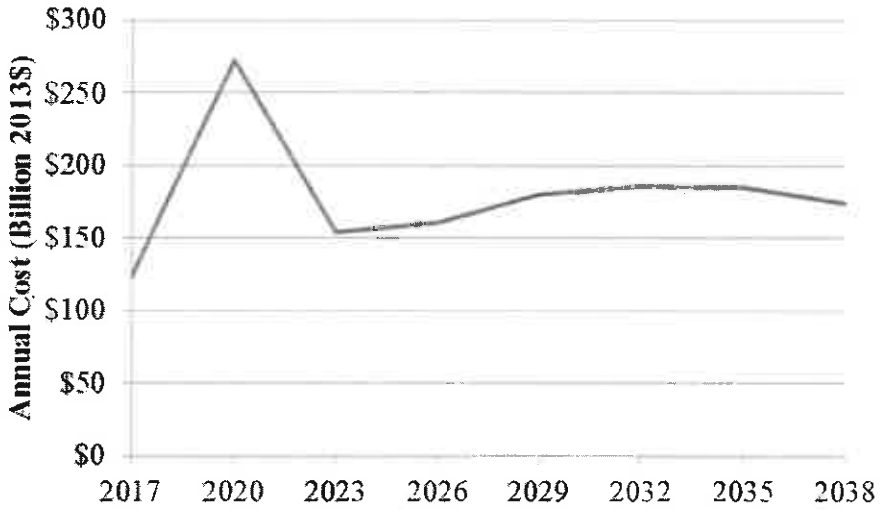
	Present Value (Billions)			Cumulative
	Capital	O&M	Total	Coal Retirements
Compliance Costs	$1,190	$1,050	$2,240	101 GW

Notes: Present value is from 2017 through 2040, discounted at a 5% real discount rate.
Cumulative coal retirements are incremental to baseline. These retirements are primarily due to assumed emission control measures but may also include indirect electric sector impacts of the ozone standards.
Source: NERA calculations as explained in text

[13] The annualized cost of $90 billion in 2020 for EPA's hybrid cost calculation with the middle slope parameter has been converted to a present value over 20 years using a real annual discount rate of 5%, converted from 2006 dollars to 2013 dollars, and calculated as of 2014. Note that there are many differences in the EPA and NERA calculations so this figure is only designed to provide a rough comparison.

Allocating the estimated capital costs to spending in years prior to each state's projected compliance deadline, and allocating O&M costs to years after the respective compliance deadlines, Figure S-6 shows the pattern of annual compliance spending across all states (except for the endogenously-determined costs of coal unit idling.)

Figure S-6: Potential Annual U.S. Compliance Spending Costs for 60 ppb Ozone Standard



Notes: Figure does not include compliance costs associated control measures in the electric power sector (scrappage of coal-fired power plants), which are modeled in N_{ew}ERA.
Source: NERA calculations as explained in text

As was the case for EPA's 2008-2010 analyses, a large portion of our estimated compliance costs are for control actions that EPA has yet to identify. As we have shown, there are ways to identify what types of controls would be needed and to develop evidence-based estimates of those types of controls. We emphasize that EPA should more fully identify the likely control measures and their costs so that the costs and economic impacts of the rule can be estimated with more confidence. Our analysis also identified the need for updated information on the identify of potential non-attainment areas and the emission reductions required to achieve compliance—additional information gaps that EPA needs to fill in order to allow for a more reliable approximation of compliance costs than it produced in 2008-2010.

Market and Macroeconomic Impacts

The prior section explains how one can develop a reasoned estimate of the resource costs of attaining a tightened ozone NAAQS and how those costs are imposed on various sectors (including households). Developing a full picture of how those costs ultimately would affect various businesses and households requires an "economic impact analysis" that takes into account the complicated interactions in the economy. According to NERA's model, changes in costs for the various sectors directly affected by the tightened ozone standard can lead to changes

in sector prices, which affect all consumers, whether households or other businesses. These regulatory costs—and the price changes they can induce—can lead to changes in the ways that sectors produce their outputs, which affects demand from existing suppliers as well as from other potential suppliers. The net effects of the compliance costs thus are potential changes in outputs and prices, with resulting effects on businesses as well as on households as workers and consumers. These impacts on businesses and households can differ dramatically by geography.

To assess these economic impacts of a regulation, the estimated compliance costs can be input into a model that includes linkages of how the directly regulated sector(s) interact with other parts of the economy. In addition to incorporating higher costs on sectors that reduce emissions, the model scenario should also reflect other types of constraints on choices or actions that the policy may impose. For example, if the regulation means that businesses cannot expand their production in certain locations, this constraint can affect the nature and location of a policy's economic impacts.

In the case of a regulation expected to have very large overall costs and to affect the costs of many sectors, a *full-economy*, macroeconomic model is needed to properly assess the overall impacts of compliance costs and other regulatory constraints. The tightened ozone NAAQS is such a situation. This study thus uses a detailed macroeconomic model to evaluate the economic impacts of a 60 ppb ozone standard.

Methodology for Analyzing Economic Impacts

We use NERA's N_{ew}ERA macroeconomic model to develop estimates of the potential macroeconomic impacts on the U.S. economy of our estimates of compliance costs for attaining a 60 ppb ozone standard. The capital costs are incurred from 2017 until 2036 (the last projected compliance date, for extreme areas), while O&M costs are incurred for all years after compliance. Our economic impact analysis includes the effects of costs incurred through 2040.

N_{ew}ERA is an economy-wide integrated energy and economic model that includes a bottom-up, unit-specific representation of the electric sector, as well as a representation of all other sectors of the economy and households. It assesses, on an integrated basis, the effects of major policies on individual sectors as well as the overall economy. It has substantial detail for all of the energy sources used by the economy, with separate sectors for coal production, crude oil extraction, electricity generation, refined petroleum products, and natural gas production. The model performs its analysis with regional detail. As discussed above, this particular analysis uses state-specific cost inputs, and N_{ew}ERA has been run to assess state-specific economic impacts. Appendix A provides a detailed description of the N_{ew}ERA model.

The analysis requires a baseline forecast that projects economic outcomes in the absence of the incremental spending to attain the tighter ozone NAAQS. For this study, N_{ew}ERA's baseline conditions were calibrated to reflect projections developed by Federal government agencies, notably the Energy Information Administration (EIA) as defined in its *Annual Energy Outlook*

2014 (*AEO 2014*) Reference case. This baseline includes the effects of environmental regulations that have already been promulgated as well as other factors that lead to changes over time in the U.S. economy and the various sectors.

Potential Impacts on the U.S. Economy and U.S. Households

The potential costs we estimated for a 60 ppb ozone standard are projected to have substantial impacts on the U.S. economy and U.S. households. Figure S-7 shows the potential macroeconomic effects as measured by gross domestic product (GDP) and U.S. household consumption. The 60 ppb ozone standard is projected to reduce GDP from the baseline levels by about $3.4 trillion on a present value basis (as of 2014) and by $270 billion per year on a levelized average basis (spread evenly over years but retaining the same present value) over the period from 2017 through 2040. Average annual household consumption would be reduced by about $1,570 per household per year.

Figure S-7: Potential Impacts of 60 ppb Ozone Standard on U.S. Gross Domestic Product and Household Consumption

	Annualized	Present Value
GDP Loss (Billions of 2013$)	$270/year	$3,390
Consumption Loss per Household (2013$)	$1,570/year	N/A

Notes: Present value is from 2017 through 2040, discounted at a 5% real discount rate. Consumption per household is an annualized (or levelized) value calculated using a 5% real discount rate.
Source: NERA calculations as explained in text

Figure S-8 focuses on several dimensions of projected impacts on income from labor ("worker income") as a result of the 60 ppb ozone standard. The projected impacts of the emissions reduction costs on labor income are substantial. Relative to baseline levels, real wages decline by about 1.2% on average over the period and labor income declines by about 1.9% on average, resulting in job-equivalent losses that average about 2.9 million job-equivalents. (Job-equivalents are defined as the change in labor income divided by the annual baseline income for the average job (see Figure S-8)). A loss of one job-equivalent does not necessarily mean one fewer employed person—it may be manifested as a combination of fewer people working and less income per worker. However, this measure allows us to express employment-related impacts in terms of an equivalent number of employees earning the average prevailing wage.[14] These are the *net* effects on labor and include the positive benefits of increased labor demand in sectors providing pollution control equipment and technologies.

[14] The N_{ew}ERA model, like many other similar economic models, does not develop projections of unemployment rates or layoffs associated with reductions in labor income. Modeling such largely transitional phenomena requires a different type of modeling methodology; our methodology considers only the long-run, equilibrium impact levels.

Figure S-8: Potential Impacts of 60 ppb Ozone Standard on Labor

	Avg.
Baseline Annual Job-Equivalents (millions)	156
60 ppb Case:	
Real Wage Rate (% Change from Baseline)	-1.2%
Change in Labor Income (% Change from Baseline)	-1.9%
Job-Equivalents (Change from Baseline, millions)	-2.9

Notes: Average (Avg.) is the simple average over 20172040. "Job-equivalents" is defined as total labor income change divided by the average annual income per job. This measure does not represent a projection of numbers of workers that may need to change jobs and/or be unemployed, as some or all of the loss could be spread across workers who remain employed

Source: NERA calculations as explained in text

Potential Effects on U.S. Energy Prices

Emissions reduction costs of a 60 ppb ozone standard also would have substantial potential impacts on U.S. energy sectors, largely because the more stringent ozone standard is projected to lead to the premature retirement of additional coal-fired power plants. Figure S-9 shows average energy price projections under the baseline and the 60 ppb ozone standard. The average delivered residential electricity price is projected to increase by an average of 3.3% over the period from 2017 through 2040. Henry Hub natural gas prices would increase by an average of 9.9% in the same time period, while delivered residential natural gas prices would increase by an average of 7.3%. Part of the increase in delivered natural gas prices reflects the increase in pipeline costs due to control costs for reductions in NO_X emissions in the pipeline system that would be recovered through tariff rates.

Figure S-9: Potential Impacts of a 60 ppb Ozone Standard on Energy Prices

		Avg. Baseline	Avg. 60 ppb Case	Change	% Change
Henry Hub Natural Gas	$/MMBtu	$6.02	$6.65	$0.63	9.9%
Natural Gas Delivered (Residential)	$/MMBtu	$13.77	$14.79	$1.02	7.3%
Natural Gas Delivered (Industrial)	$/MMBtu	$8.43	$9.49	$1.06	12%
Gasoline	$/gallon	$3.56	$3.57	$0.01	0.4%
Electricity (Residential)	¢/kWh	14.5¢	14.9¢	0.5¢	3.3%
Electricity (Industrial)	¢/kWh	9.4¢	9.9¢	0.5¢	5.5%

Notes: Average is the simple average over 2017-2040.

Source: NERA calculations as explained in text

Potential Effects on U.S. Sectors and Regions

All sectors of the economy would be affected by a 60 ppb ozone standard, both directly through increased emissions control costs and indirectly through impacts on affected entities' customers and/or suppliers. There are noticeable differences across sectors, however. Figure S-10 and Figure S-11 show the estimated changes in output for the non-energy and energy sectors of the economy, respectively, due to the emissions reduction costs of a 60 ppb ozone standard.

Figure S-10: Potential Impacts of 60 ppb Ozone Standard on Output of Non-Energy Sectors (Percentage Changes from Baseline)

	Agriculture	Commercial/ Services	Manufacturing	Commercial Transportation	Commercial Trucking
Average (2017-2040)	-2.2%	-0.9%	-0.6%	-1.9%	-1.1%

Notes: Values are the simple average of percentage change over 2017-2040.
Source: NERA calculations as explained in text

Figure S-11: Potential Impacts of a 60 ppb Ozone Standard on Output of Energy Sectors (Percentage Changes from Baseline)

	Coal	Natural Gas	Refining	Crude Oil	Electricity
Average (2017-2040)	-52%	9.2%	-1.8%	-0.1%	-3.1%

Notes: Values are the simple average of percentage change over 2017-2040.
Source: NERA calculations as explained in text

Figure S-12 shows the average annual change in consumption per household for individual N_{ew}ERA regions. A region's attainment costs and its sectoral output mix determine to a large extent whether a region fares better or worse than the U.S. average, but all regions would experience lower household consumption.

Figure S-12: Potential Impacts of a 60 ppb Ozone Standard on Annual Consumption per Household by Region

Region	
Arizona and Mountain States	-$690
California	-$2,910
Florida	-$450
Mid-America	-$850
Mid-Atlantic	-$2,520
Mississippi Valley	-$1,550
New York/New England	-$2,490
Pacific Northwest	-$730
Southeast	-$1,060
Texas, Oklahoma, Louisiana	-$1,070
Upper Midwest	-$1,770
U.S.	**-$1,570**

Note: Values are the levelized average over 2017-2040, annualized using a 5% real discount rate. Maps of N_{ew}ERA regions are provided in the report body and Appendix A.
Source: NERA calculations as explained in text

Sensitivity Case with Limits on Natural Gas Production

The above results assume that the U.S. energy sectors would be able to increase production with no permitting delays or constraints (other than the higher production cost associated with emissions controls) in order to meet increased natural gas demand associated with ozone NAAQS attainment actions. This assumption allows for a large projected increase in U.S. natural gas production. However, natural gas producers in areas that become nonattainment under a tighter ozone standard might face new requirements – such as the need to obtain air permits as well as emissions reduction credits ("offsets") for NO_X and/or VOCs – in order to develop new wells. Whether such permitting requirements will be applied to new oil and gas extraction nationally is a policy question that is in a state of flux at present; but some areas of the country already have these requirements and there are pressures for the EPA to make it a uniform requirement. Moreover, expansion of natural gas output will require additional gas processing facilities, which are already subject to the offsetting requirement if located in nonattainment areas. Obtaining offsets may be difficult and/or costly, particularly in relatively rural areas that are likely to face nonattainment issues that until now have been mainly faced by urban areas. Such rural areas will have few industrial emissions sources to create offset supply, so that a potential requirement of new sources in nonattainment areas to purchase offsets may become a substantial hindrance to growth. If such barriers to new well development do emerge, the projected economic impacts of a 60 ppb ozone NAAQS could be substantially increased.

To explore the ways that a constraint on new well development could change the economic impacts of a 60 ppb ozone standard, we developed a natural gas production sensitivity case. For this case we used the same emissions reduction cost inputs as in the 60 ppb case, but we also assumed that total U.S. natural gas production would not increase beyond its 2020 level as modeled in the 60 ppb scenario.[15] We project that lower 48 U.S. natural gas production would be 28.9 quadrillion Btus ("quads") in 2020 under a 60 ppb ozone standard without any natural gas constraints, so we limited lower 48 U.S. natural gas production to 28.9 quads after 2020 in the sensitivity case. Note that limits on natural gas production may also affect crude oil production, but we have not attempted to evaluate this possibility.

Figure S-13 shows the potential effects of a 60 ppb ozone standard in the natural gas production sensitivity case (which includes the effects of reduced natural gas availability as well as the estimated compliance costs) on the U.S. economy as measured by GDP and U.S. household consumption. The 60 ppb ozone standard with assumed natural gas production limits is projected to reduce GDP from the baseline levels by about $360 billion per year on a levelized average basis and by $4.5 trillion on a present value basis (as of 2014) over the period from 2017 through 2040, which is about 30% higher than for the 60 ppb case without natural gas production limitations. Average household consumption would be reduced by about $2,040 per household per year on average over the period.

Figure S-13: Potential Impacts of a 60 ppb Ozone Standard on U.S. Gross Domestic Product and Household Consumption (Sensitivity Case)

	Annualized	Present Value
GDP Loss (Billions of 2013$)	$360/year	$4,480
Consumption Loss per Household (2013$)	$2,040/year	N/A

Notes: Present value is from 2017 through 2040, discounted at a 5% real discount rate
Source: NERA calculations as explained in text

[15] This scenario constrained natural gas production to be at or below 2020 levels throughout the model horizon, but this policy does not mean that no new wells could be drilled. If new wells were prohibited after 2020, U.S. natural gas production would actually start to decline after 2020, rather than hold steady as this case assumes.

Figure S-14 shows potential labor impacts as a result of the 60 ppb ozone standard with natural gas production constraints. In this sensitivity case, labor income would decrease (relative to baseline future levels) by an average of 2.7% over the period from 2017 through 2040, and in job-equivalents this would imply an average annual loss of about 4.3 million job-equivalents.

Figure S-14: Potential Impacts of a 60 ppb Ozone Standard on Labor (Sensitivity Case)

	Avg.
Baseline Job-Equivalents (millions)	155.7
Sensitivity Case:	
Real Wage Rate (% Change from Baseline)	-2.0%
Change in Labor Income (% Change from Baseline)	-2.7%
Job-Equivalents (Change from Baseline, millions)	-4.3

Notes: Average is the simple average over 2017-2040. "Job-equivalents" is defined as total labor income change divided by the average annual income per job. This value does not represent a projection of numbers of workers that may need to change jobs and/or be unemployed, as some or all of it could be spread across workers who remain employed.
Source: NERA calculations as explained in text

Figure S-15 shows potential energy prices under the baseline and the 60 ppb ozone standard with natural gas production constraints. In this sensitivity case, the average delivered residential electricity price is projected to increase by an average of 15% over the period from 2017 through 2040. Henry Hub natural gas prices would increase by an average of 66% in the same time period, while delivered residential natural gas prices would increase by an average of 32%.

Figure S-15: Potential Impacts of a 60 ppb Ozone Standard on Energy Prices (Sensitivity Case)

		Avg. Baseline	Avg. Sensitivity	Change	% Change
Henry Hub Natural Gas	$/MMBtu	$6.02	$9.97	$3.95	66%
Natural Gas Delivered (Residential)	$/MMBtu	$13.77	$18.16	$4.39	32%
Natural Gas Delivered (Industrial)	$/MMBtu	$8.43	$12.79	$4.36	52%
Gasoline	$/gallon	$3.56	$3.60	$0.04	1.3%
Electricity (Residential)	¢/kWh	14.5¢	16.6¢	2.1¢	15%
Electricity (Industrial)	¢/kWh	9.4¢	11.6¢	2.2¢	23%

Notes: Average is the simple average over 2017-2040.
Source: NERA calculations as explained in text

Figure S-16 and Figure S-17 show the estimated changes in output for the various sectors of the economy due to a 60 ppb ozone standard under the sensitivity case with natural gas production constraints. This case leads to substantial reductions in natural gas output relative to the baseline.

Figure S-16: Potential Impacts of a 60 ppb Ozone Standard on Output of Non-Energy Sectors (Percentage Changes from Baseline) (Sensitivity Case)

	Agriculture	Commercial/ Services	Manufacturing	Commercial Transportation	Commercial Trucking
Average (2017-2040)	-2.7%	-1.2%	-1.3%	-2.4%	-1.5%

Notes: Values are the simple average of percentage change over 2017-2040.
Source: NERA calculations as explained in text

Figure S-17: Potential Impacts of a 60 ppb Ozone Standard on Output of Energy Sectors (Percentage Changes from Baseline) (Sensitivity Case)

	Coal	Natural Gas	Refining	Crude Oil	Electricity
Average (2017-2040)	-52%	-11%	-2.3%	0.2%	-9.7%

Notes: Values are the simple average of percentage change over 2017-2040.
Source: NERA calculations as explained in text

Recommendations for Forthcoming Ozone Regulatory Impact Analysis

The large potential costs and economic impacts reported in this study—along with the substantial uncertainties involved in their estimation—suggest two major recommendations for EPA as it prepares the RIA for its forthcoming ozone proposal:

1. EPA should develop analyses of the overall costs and economy-wide impacts of more stringent ozone standards; and

2. EPA should provide updated information on critical parameters, including the potential permitting barriers on oil and gas production in nonattainment areas as well as updated and expanded estimates of the emission reductions and costs required to achieve alternative ozone standards.

We have developed estimates of the potential impacts of a 60 ppb ozone standard on the U.S. economy and on U.S. households using the best available information on emissions and controls, including the impacts of a sensitivity case in which we assume that U.S. natural gas production would be constrained after 2020 as a result of the ozone standard. It will be important for EPA to

provide these types of assessments based upon its estimates of compliance costs and resulting impacts on the economy of alternative ozone standards. There are sound reasons to expect a revised ozone standard to be very costly to attain and these costs would likely have major adverse macroeconomic impacts.

It is important that the attainment cost and macroeconomic impact estimates be based upon reliable information. Our analyses uncovered numerous gaps that EPA should fill as it develops its RIA. Perhaps the most important gaps are the identity of control options and their costs to achieve the emissions reductions needed for attainment, although it is important to develop updated information on the specific emission reduction requirements as well. The bulk of compliance costs to meet a 60 ppb standard in EPA's 2008-2010 analyses are based upon "unknown" controls, *i.e.*, controls that are not attributed to particular control technologies or even to particular sectors. We have developed estimates of these "unknown" costs based upon an assessment of the available information, particularly the sources of the emissions remaining by 2018 that would need to be reduced to attain a tighter ozone NAAQS, as well as on costs of potential retirement of coal units and potential reductions from mobile sources and where they might fit in a marginal cost curve. But it would be important for EPA to update and expand its compliance cost information to provide a more comprehensive assessment of emission control options and compliance costs. Moreover, our sensitivity analysis including natural gas production constraints suggests the importance of this issue and thus the need for EPA to evaluate the potential impacts of a tighter ozone standard on domestic natural gas and crude oil production.

I. INTRODUCTION

This report provides estimates of the potential impacts on the U.S. economy of a more stringent national ambient air quality standard (NAAQS) for ozone and recommends updated and expanded information the U.S. Environmental Protection Agency (EPA) should provide when it issues a proposal. The analysis is based upon the most up-to-date information on projected ozone precursor emissions as well as the best available information on control costs. We use our integrated energy-economic model, N_{ew}ERA, to estimate the potential macroeconomic effects of complying with a more stringent standard. The limited information now available to assess these compliance costs and economic impacts provides a compelling argument that if EPA proposes a stricter standard, it needs to develop more complete data and analysis as part of its forthcoming proposal. We refer to our estimates as potential costs and economic impacts to reflect the substantial uncertainties in the underlying emission reduction and cost information.

A. Background

1. Policy Background

The U.S. Environmental Protection Agency (EPA) has responsibility under Sections 108 and 109 of the Clean Air Act to establish, to review and to revise (as appropriate) a primary NAAQS that protects the nation's public health with an "adequate margin of safety." This assessment is made by the EPA Administrator based upon a review of various EPA assessments as well as review of advice from the Clean Air Scientific Advisory Committee (CASAC). Once a national standard is revised, states have the responsibility to develop State Implementation Plans (SIPs), documents that describe how the states will ensure that regions within their jurisdiction will attain and maintain the standard. States typically are given attainment deadlines that vary depending upon the severity of nonattainment. EPA has set NAAQS for six principal pollutants.

The Clean Air Act instructs EPA to review the NAAQS every five years. The EPA in March 2008 set an ozone standard of 75 parts per billion (ppb), lowering the standard from 0.08 parts per million (effectively 84 parts per billion (ppb) due to rounding conventions). In 2010, EPA reconsidered the ozone standard and evaluated lower potential standards, including 60 ppb. EPA currently is reviewing the ozone NAAQS, including developing various assessment materials and obtaining advice from the CASAC. EPA has stated its intention to consider tightening the standard to as low as 60 ppb in its most recent draft policy assessment. Our study thus evaluates a new ozone standard of 60 ppb, one value that seems likely to be included when EPA issues a proposed rule.

Although EPA's review of the NAAQS for ozone is under way, EPA has not released any new ozone compliance cost estimates since its 2008-2010 analyses.[16] The Agency has issued some

[16] "EPA's 2008-2010 ozone analyses" refers to information in EPA's 2008 regulatory impact analysis (RIA) for the ozone NAAQS, including information on baseline future conditions and ozone standards of 84 and 75 ppb (EPA

updated information on projected baseline emissions, however. In addition, there is updated information on monitored ozone concentrations that indicates the air quality regions and states most likely to be designated in nonattainment with a 60 ppb standard. The updated information allows us to develop estimates of the emissions reductions that would be required for these states to come into attainment, which we use to develop estimates of the costs of such a tightened NAAQS for ozone. The information EPA has currently made available is limited and, indeed, this analysis is provided in part to illustrate the approach and types of data that EPA should develop to provide a sound understanding of the economic impacts of a new ozone NAAQS when it releases its proposal. The approach and data development that are needed will be the same whether EPA chooses to propose a standard of 60 ppb, as we analyze here, or some other level.

2. Ozone Concentrations

a. EPA Recent Historical Information

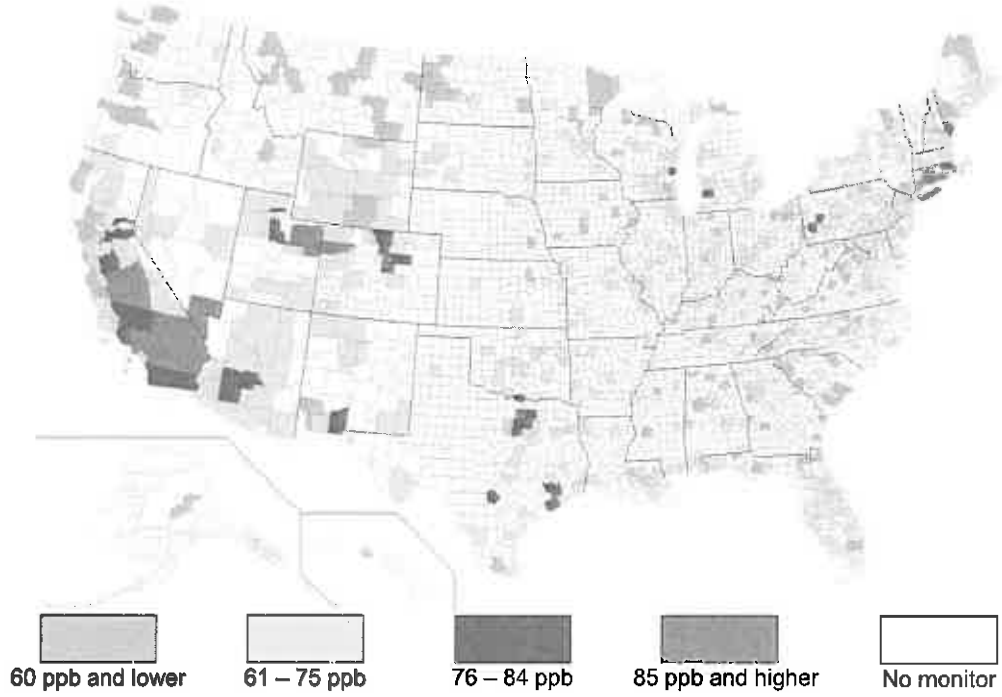
EPA uses a network of air quality monitors to measure concentrations of ozone and other air pollution across the country. Figure 1 shows ozone readings for 2013 (measured as the fourth-highest level over an eight-hour period) in the 781 counties with ozone monitors that year; note that about three quarters of all U.S. counties did not have ozone monitors. Although some areas of the country were below (or at) 60 ppb, most areas with ozone monitors were above that level. Many states had areas with 2013 ozone concentrations above the 2008 standard of 75 ppb, and six states (California, Colorado, Connecticut, Nevada, Texas, and Utah) had areas above the 1997 standard of 80 ppb (effectively 84 ppb due to rounding conventions). As discussed above, states develop SIPs to bring nonattainment areas into compliance.

b. EPA Projections from 2008 Ozone NAAQS Review

Figure 2 shows EPA's projections from the 2008 ozone NAAQS review of baseline ozone concentrations in 2020 in the 661 counties that had ozone monitors when EPA performed this analysis. As shown in the figure, almost all states were projected to have an area with ozone concentration above 60 ppb under baseline conditions in 2020. EPA has not issued new ozone projections since its 2008-2010 ozone analyses

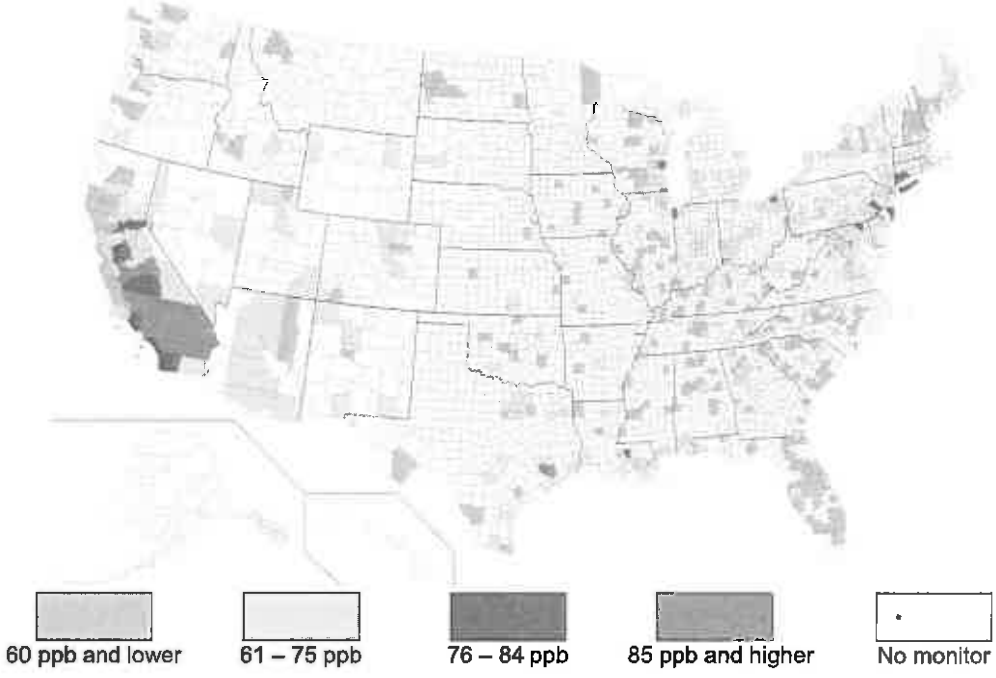
2008); EPA's 2010 supplemental regulatory impact analysis, including information on an ozone standard of 60 ppb (EPA 2010b); and data files in Docket No. EPA-HQ-OAR-2007-0225.

Figure 1: EPA 2013 Ozone Concentration Data



Source: NERA map using ozone concentration data from EPA (2014c)

Figure 2: EPA Projections of 2020 Baseline Ozone Concentrations from 2008 NAAQS Analysis



Source: NERA map using ozone concentration data in EPA (2008, pp. 3a-31 to 3a-45)

3. Energy Developments

Recent energy developments in the United States have important implications for ozone concentrations and air policy. Technological advances in oil and gas extraction (including improved horizontal drilling techniques) have significantly expanded the available resource base in recent years. Between 2005 and 2013, U.S. oil production increased 43% and U.S. natural gas production increased 34% (EIA 2014). Extraction and transportation of crude oil and natural gas produce emissions that contribute to ozone concentrations. A tighter ozone standard could introduce air permit and emissions offset requirements that might constrain new crude oil and natural gas activity in nonattainment areas. To provide an indication of the potential impacts, we performed a sensitivity case with constraints on natural gas production. Dramatic changes have occurred recently in domestic energy resources, and we provide this sensitivity analysis in part to indicate the types of data that EPA should develop when it releases its proposal in order to provide a sound understanding of the economic impacts of a more stringent ozone NAAQS.

B. Objectives of This Report

This report has two principal objectives:

1. Assess the costs and economic impacts of a 60 ppb ozone standard using the best available information from EPA and other sources; and
2. Develop recommendations for additional and updated information and analyses EPA should provide in its regulatory impact analysis (RIA) of a proposed rule, so that such assessments can be more fully evidence-based.

The first objective is predicated on the large potential significance to the U.S. economy of a more stringent ozone standard. Unlike regulations that target specific sectors, an ozone standard would directly affect virtually every sector of the economy since ozone precursors (oxides of nitrogen, or NO_X, and many types of volatile organic compounds, or VOCs) are emitted by a wide range of stationary, mobile, and area sources. Moreover, the analyses undertaken by EPA as part of its 2008-2010 ozone review make it clear that the overall costs would be very large. Indeed, as noted, the standard might result in other effects, notably potential constraints on domestic natural gas and crude oil development if nonattainment regions introduce permitting barriers or require emissions offsets to develop new wells and processing facilities. Our estimates show the potential costs and macroeconomic impacts of a more stringent ozone standard, impacts that are even greater under a sensitivity case that assumes nonattainment limits new natural gas wells.

The second objective relates to EPA's process of updating its analysis as it prepares its RIA. Our analysis reveals major gaps in data on compliance technologies and their costs and in other information. Our research puts us in a position to recommend information that EPA should develop and make available in order to provide comprehensive and reliable assessments of a more stringent ozone standard. EPA needs to provide updated data and analyses so that it can

provide an adequate understanding of the economic impacts of a tighter ozone NAAQS. Recognizing the large uncertainties and gaps of currently available information from EPA, we refer to our results as potential costs and macroeconomic impacts.

C. Report Organization

The remainder of the report is organized as follows. Section II provides information on the methodology used in our study. Section III presents the empirical results of the study. Section IV summarizes our conclusions.

II. METHODOLOGY AND COMPLIANCE COSTS

This chapter provides an overview of the methodology that we use to evaluate the effects of a 60 ppb ozone standard on the U.S. economy. The first section includes an overview of N_{ew}ERA, the model we use to develop estimates of the macroeconomic impacts, as well as information on the compliance cost inputs we develop. The second section explains our methodology for computing compliance costs. The last section describes the sensitivity case, which considers a potential impact of the ozone standard on natural gas production.

A. N_{ew}ERA Model

1. Overview of the N_{ew}ERA Model

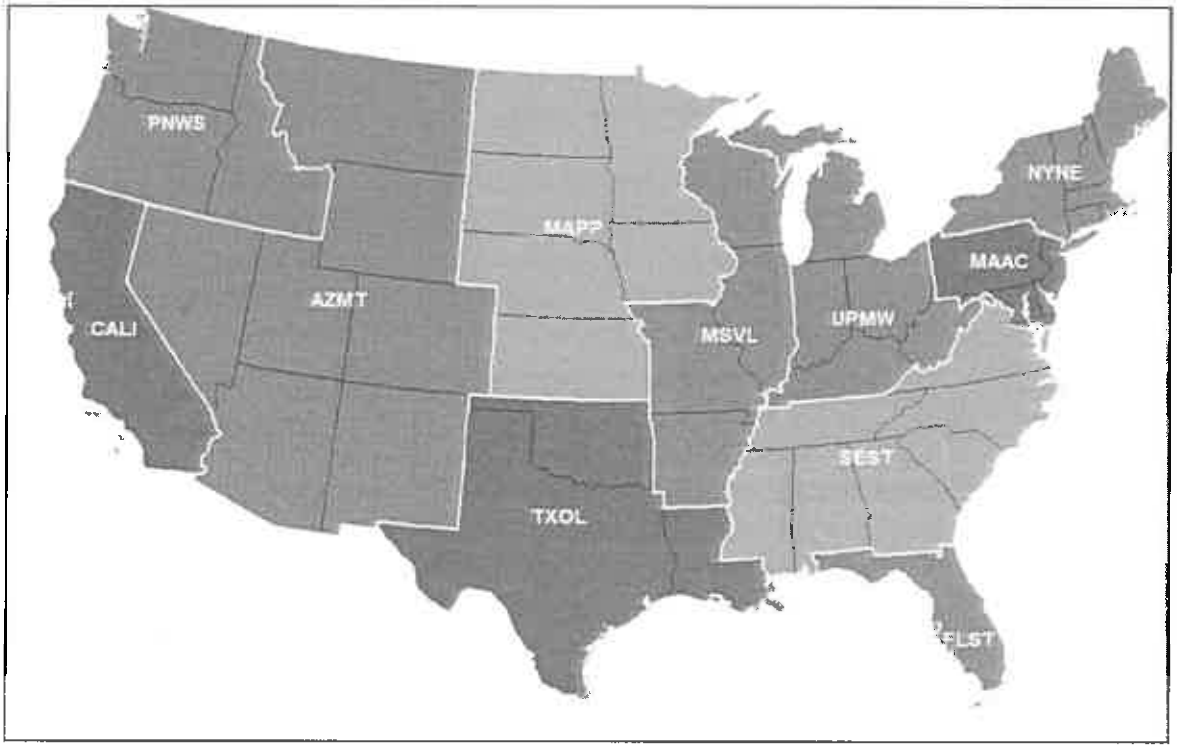
The N_{ew}ERA model is an economy-wide integrated energy and economic model that includes a bottom-up, unit-specific representation of the U.S. electricity sector, as well as a top-down representation of all other sectors of the economy and households. It assesses, on an integrated basis, the effects of major policies on individual sectors as well as the overall economy. It has substantial detail for all of the energy sources used by the economy, with separate sectors for coal, crude oil extraction, electricity generation, refined petroleum products, and natural gas production. The model performs its analysis with regional and state detail, accounting for over 30 electricity market regions and 11 macroeconomic regions in the lower 48 states for other economic activities (see Figure 3). The N_{ew}ERA model is a long-term model that includes the assumption that households and firms develop optimum decisions over the modeling period. Appendix A provides a detailed description of the N_{ew}ERA model.

We developed state-specific inputs for N_{ew}ERA and performed multiple modeling iterations to generate state-specific economic impacts. As an integrated model, N_{ew}ERA incorporates supply and demand connections among sectors of the economy. It is able to account for both the adverse effects of higher business costs and "unproductive" capital required by environmental regulations as well as the near-term gains to companies that manufacture pollution control equipment and other means of compliance with environmental regulations. For example, when sectors must incur capital costs for pollution control measures, these capital costs represent increased demand for manufacturing sectors goods that would produce the pollution controls.

2. Baseline Conditions

The baseline scenario is constructed from a version of the N_{ew}ERA model that is calibrated to the most recent *Annual Energy Outlook* (*AEO 2014*) Reference case forecast. That is, the model's parameters and key inputs such as natural gas supply/prices are first set so that if we impose the same policies as are in AEO 2014, N_{ew}ERA will produce very similar projected fuel prices and consumption.

Figure 3: N_{ew}ERA Macroeconomic Model Regions



Source: N_{EW}ERA model definitions

The N_{ew}ERA and *AEO 2014* baseline forecasts incorporate current environmental regulations, including the following major programs:

- Regional Greenhouse Gas Initiative (RGGI);[17]
- California AB 32 greenhouse gas emission program;
- SO_2 and NO_X emission programs for the electricity sector;
- Mercury emission limits under the EPA Mercury and Air Toxics Standards (MATS); and
- Recent air emission standards for new passenger cars and light duty trucks.

Neither N_{ew}ERA nor the modeling behind the *AEO 2014* forecasts constrains air emissions for compliance with current NAAQS. In particular, air emission projections from the models under baseline conditions do not achieve compliance in all areas of the country with the current ozone standard of 75 ppb. The models do not incorporate state-specific controls that might be put in

[17] *AEO 2014* did not include recent updates to make the RGGI policy more stringent over time. For consistency purposes, N_{ew}ERA also did not include these updates.

place to comply with this standard (the controls are currently under development as states continue to prepare and implement their SIPs for 75 ppb). In addition, the baseline excludes EPA's recently proposed rule to limit carbon dioxide (CO_2) emissions from the power sector because it is not part of current law. As discussed below, taking into account EPA's recent CO_2 rule would not change the major conclusions of our study.

On June 2, 2014, EPA released a proposed rule to limit CO_2 emissions from the power sector. Implementation of such a rule will almost certainly result in some amount of NO_X reduction (primarily because of a reduction in coal-fired generation needed to reduce state CO_2 emissions rates). EPA (2014d, pp. ES-6 and ES-7) estimates that the proposed power sector CO_2 rule would reduce annual NO_X emissions by approximately 300,000 to 400,000 tons (depending on regulatory option, state or regional compliance approach, and measurement year). Some of those NO_X reductions may overlap with NO_X reductions in our ozone cost analysis, and to the extent that this would occur, some of the cost estimates will be shared with the cost of the proposed CO_2 rule for the power sector. We have considered this issue and find that even if all of those overlapping costs were to be removed from our analysis, the costs and economic impacts presented in this report would not change in any meaningful degree because NO_X reductions from the power sector are estimated to be among the lowest cost-per-ton of the NO_X reductions in our ozone attainment scenario. In fact, even if all of the approximately 100 GW reduction in electric sector coal capacity (discussed in subsequent sections of this report) were treated as costless in our ozone analysis, our estimated GDP impact would only be reduced by about 8%, or about one-twelfth of the value we estimate. Moreover, our review of EPA's RIA for the power plant CO_2 rule indicates that the overlap of power sector costs is much less than 100%, as the proposed power sector CO_2 rule would lead to NO_X reductions in some areas that would not require reductions for compliance with a new 60 ppb ozone standard. Thus, uncertainty on how to attribute shared costs between the two regulations does not affect the major conclusions of this report regarding the costs and economic impacts of a tightened ozone NAAQS.

3. Modeling Years

For this analysis, we evaluate the economic implications of a 60 ppb ozone standard, with compliance in individual states staggered to reflect the degree of nonattainment (as discussed below). We model results for three-year periods beginning with 2017. Thus, we present results for 2017, 2020, 2023, 2026, 2029, 2032, 2035, and 2038. Each model year represents an average of three years, the stated year and the next two years; for example, 2017 represents the average of 2017 through 2019. These annual results are used to calculate present values for 2017 through 2040, as of 2014.

B. Compliance Cost Inputs to N_{ew}ERA

This section summarizes the steps we take to develop state- and sector-specific compliance cost estimates for an ozone standard of 60 ppb. The methodology consists of the following three major steps:

1. Estimate the NO_X reductions needed to achieve 60 ppb in each state;
2. Estimate the costs of achieving the required NO_X reductions in each state; and
3. Allocate compliance costs to years and N_{ew}ERA sectors.

The steps are based upon the most up-to-date information available from EPA, including information from the 2008 RIA as well as additional materials that EPA has released since then. We focus on NO_X emissions and emission reductions in our analysis because EPA indicates that NO_X is the critical precursor for ozone formation in most areas of the country, particularly for a tighter new standard of 60 ppb (EPA 2010, pp. S2-3 and S2-14); our compliance cost estimates, however, do include EPA's estimates of the costs of reducing volatile organic compounds (VOCs), the other major ozone precursor. We describe each of these major steps in the subsections below. Appendices B, C, and D provide details on the methodology and data and present state-specific information.

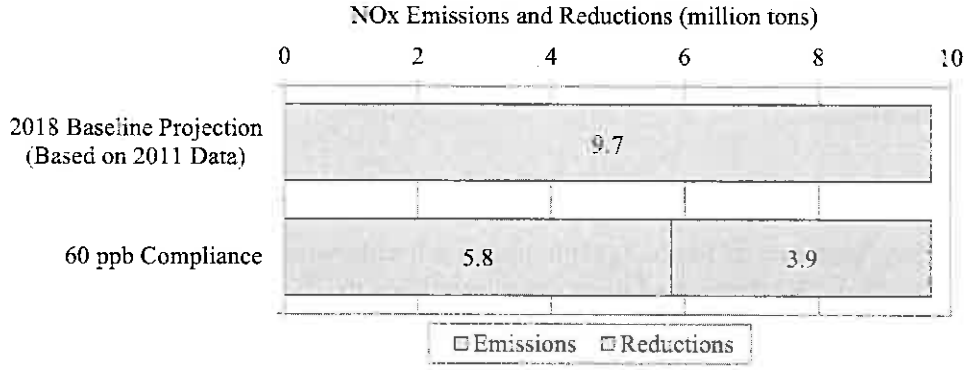
1. Step 1: Determine Necessary NO_X Reductions by State

The first step is to develop estimates of the NO_X reductions each state would need in order to meet a 60 ppb ozone standard. We developed these estimates using two sources of data: (1) recent EPA projections of future baseline NO_X emissions for 2018; and (2) estimates from EPA's 2008 RIA of the level of NO_X emissions consistent with meeting a 60 ppb standard.

The EPA in December 2013 released detailed estimates of projected 2018 NO_X emissions using its most recent projection platform, which is based upon historical emissions data from 2011 (2014a). In its 2008 RIA, EPA had developed estimates of projected 2020 NO_X emissions as well as estimates of the reductions that would be required to achieve a 60 ppb standard (among other standards) in 2020; the difference between these two values represents a set of estimates of the state-by-state emissions that would be consistent with national compliance with a 60 ppb standard ("compliance emissions"). The reduction requirements modeled in EPA's 2008-2010 ozone review (which we use to determine state-level compliance emissions) extended beyond the limited number of counties with ozone monitors shown in Figure 1 and Figure 2 since EPA assumed that non-monitored areas near nonattaining counties would also be required to reduce their NO_X emissions to achieve attainment. Note, however, that the number of nonattainment counties may be greater than EPA's existing monitoring sites, and thus the required emission reductions, compliance costs and economic impacts may be greater than based upon the current EPA monitoring information.

For each state, we calculated the NO_X emission reductions that would be required by subtracting the compliance emissions from the projected 2018 baseline emissions. Figure 4 shows national estimates of baseline emissions, compliance emissions, and required reductions. Based on information in the 2008-2010 EPA ozone review, U.S. NO_X emissions would have to be reduced to about 5.8 million tons to meet a 60 ppb standard in 2020. National NO_X emissions have been decreasing in recent years, from about 25.2 million tons in 1990, to 16.8 million tons in 2008, and to 12.9 million tons in 2013 (EPA 2014b). EPA presently is projecting that NO_X emissions will be reduced to 9.7 million tons by 2018 (supplemented with EGU baseline emission projections from N_{ew}ERA) due to existing rules and regulations (EPA 2014a), some of which have not yet been fully implemented and will carry with them additional compliance costs on top of the compliance costs estimated in this study. These emissions estimates mean that in 2018 another 3.9 million tons of NO_X will need to be reduced to get to the 60 ppb standard nationally (this reduction is equivalent to about 40% of baseline NO_X emissions for 2018).

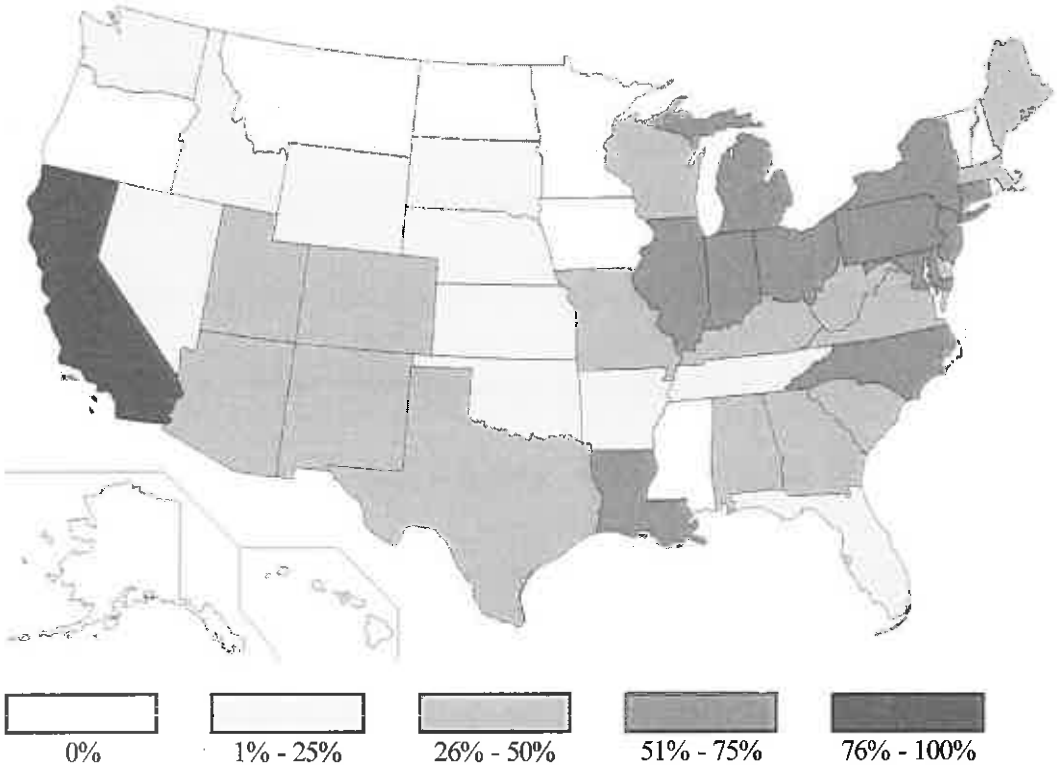
Figure 4: National 2018 Baseline NO_X Emissions, 60 ppb Compliance NO_X Emissions, and 60 ppb NO_X Required Emission Reductions



Note: NO_X emissions include only anthropogenic sources (*i.e.*, exclude fires and biogenic sources).
Source: NERA calculations based on EPA (2014a) and other inputs as explained in Appendix B

These values are stated on a national basis, but the actual emissions control requirements will be concentrated around projected nonattainment areas. Moreover, the emissions reductions needed in any particular nonattainment area will vary depending on the severity of its exceedance of the NAAQS level. Figure 5 summarizes state NO_X emission reductions for 60 ppb compliance (expressed as a percentage reduction from EPA 2018 baseline emission projections). Our analysis projects that 40 states would have at least one nonattainment area in 2018 under a 60 ppb ozone NAAQS and thus will have to reduce NO_X emissions. We also find that 12 of these 40 states will need to reduce their NO_X emissions by between 50% and 80% from their already-reduced levels projected in 2018 to attain the standard. Appendix B presents detailed state information on projected 2018 emissions, compliance emissions, and emission reductions required to achieve a 60 ppb ozone standard.

Figure 5: Necessary NO_X Emission Reductions for 60 ppb (Percentage Reduction from Baseline 2018 NO_X Emission Projections)



Note: Map shows necessary percentage reductions below 2018 baseline NO_X emission projections. States with 0% reductions would comply with a new ozone standard of 60 ppb in 2018 based on this analysis.

Source: NERA calculations as explained in text and Appendix B

2. Step 2: Determine NO_X Compliance Costs by State

The next major step in our analysis is to develop estimates of the compliance costs that would be incurred in each state in order to obtain the required NO_X emission reductions. We developed compliance cost estimates using the most recent data available from EPA, supplemented by estimates for controls not considered by EPA, and judgments about the nature of the marginal cost curve (*i.e.*, the relationship between the marginal costs of reducing NO_X emissions and the extent of NO_X reductions). As discussed below, we considered it important to develop an evidence-based assessment of the costs EPA described as "unknown" in their 2008-2010 analysis and for which they developed a largely arbitrary method of estimation, in part to provide a template for the type of detailed cost analysis EPA should do as part of its forthcoming RIA.

a. EPA Costs for "Known" and "Unknown" Controls

In its 2008-2010 ozone analyses, EPA presented state-specific information on "known" controls. These "known" controls represent specific control measures that regions and states could use to reduce their NO_X emissions. EPA identified controls from five categories of emission sources:

(1) Electric generating units (EGUs);

(2) Non-EGU point sources, such as industrial boilers, cement kilns, and petroleum refineries;

(3) Area sources, such as dry cleaners, commercial buildings, and residential buildings;

(4) Onroad mobile, such as passenger cars, light-duty trucks, and heavy-duty trucks; and

(5) Nonroad mobile, such as locomotives, aircraft, marine vessels, construction equipment, and agricultural equipment.

The RIAs and their supporting documentation include estimates of emission reductions and annualized costs by facility or state for "known" controls in these sectors.

After applying all "known" controls to the five categories of emission sources, EPA found that certain areas would not meet some of the alternative ozone standards, including 60 ppb. In these cases, EPA assumed that the areas would achieve the standards through installation of "unknown" controls. To estimate the potential total costs of alternative ozone standards, EPA developed a marginal cost curve (*i.e.*, curve showing relationship between the incremental or "marginal" cost of controls and NO_X emissions reduced) starting with the "known" controls in order from the lowest to the highest cost per ton reduced. The curve used a "slope" parameter based on "known" control costs and an arbitrary assumption on the maximum cost per ton in order to extend the curve beyond "known" controls to include the costs of "unknown" controls. EPA's methodology and cost estimates are discussed in Appendix C.

We used EPA's estimates of "known" controls as the starting points for the estimated marginal cost curves we developed for each state. For the United States as a whole, the "known" controls are projected to reduce NO_X emissions by about 1.3 million tons per year, or about one-third of the reductions required to achieve the 60 ppb ozone standard.

The following table summarizes the EPA "known" controls.

Figure 6: National Summary of EPA "Known" NO_X Controls (tons of reduction)

Point (Non-EGU)	**825,400**
Selective catalytic reduction (SCR) without low-NOx burner (LNB)	466,800
Low-emission combustion (for internal combustion engines)	82,000
Selective catalytic reduction (SCR) and low-NOx burner (LNB)	80,800
Non-selective catalytic reduction (NSCR)	70,300
Selective non-catalytic reduction (SNCR)	61,400
OXY-firing (for glass manufacturers)	33,800
Low-NOx burner (LNB) without selective catalytic reduction (SCR)	20,700
Biosolid injection (for cement kilns)	8,200
Other	1,300
Area	**27,800**
Low-NOx water and space heaters (for commercial buildings)	14,000
Low-NOx burner (LNB)	12,800
Switch to low-sulfur fuel (for residential buildings)	1,000
Onroad Mobile	**256,100**
Retrofit heavy-duty diesel with selective catalytic reduction (SCR)	137,700
Continuous inspection and maintenance	27,800
Eliminate long-duration idling	10,500
Commuting programs	4,400
Low Reid Vapor Pressure	1,000
Unspecified	74,800
Nonroad Mobile	**45,000**
Retrofit heavy-duty diesel with selective catalytic reduction (SCR)	45,000
No Details (Some Omissions in EPA Data for CA and TX)	**130,100**
Total	**1,284,400**

Note: Totals may not equal sum of rows due to independent rounding.
Source: NERA calculations as explained in text

These controls have a range of marginal costs up to $23,000 per ton (2006 dollars), the maximum assumed by EPA in its 2008-2010 analysis (EPA 2008, p. 5-3). For some states, these known controls were sufficient to achieve the NO_X emission reductions required to achieve a 60 ppb standard—and, indeed, as shown in Figure 5 above, eight states (excluding Alaska, Hawaii, and the District of Columbia) did not require any emission reductions.[18] But the bulk of states are projected to require emission reductions beyond the known controls, and in some states the additional emission reductions are very substantial. EPA's estimates of the costs of "unknown" controls provided in the 2008-2010 RIAs were based upon estimates of "known" control costs and an arbitrary maximum cost per ton, rather than on estimates of the additional controls that would be required.

b. NERA Cost Estimates for "Unknown" Controls

We developed estimates of the potential costs of "unknown" controls by developing answers to the following three questions

- What categories of emission sources would be potentially available to achieve these additional 2.6 million tons of "unknown" NO_X reductions?
- What types of control strategies would likely be used for these "unknown" NO_X emission reductions?
- What would be the costs of these "unknown" controls?

To address the first two of the above questions, we disaggregated EPA's estimates of 2018 baseline emissions down to the five categories that EPA identifies in its emissions inventories, and we allocated the reductions from EPA's list of known controls to these categories.[19] This information gives insights on what types of control strategies might be available to obtain further "unknown" reductions.

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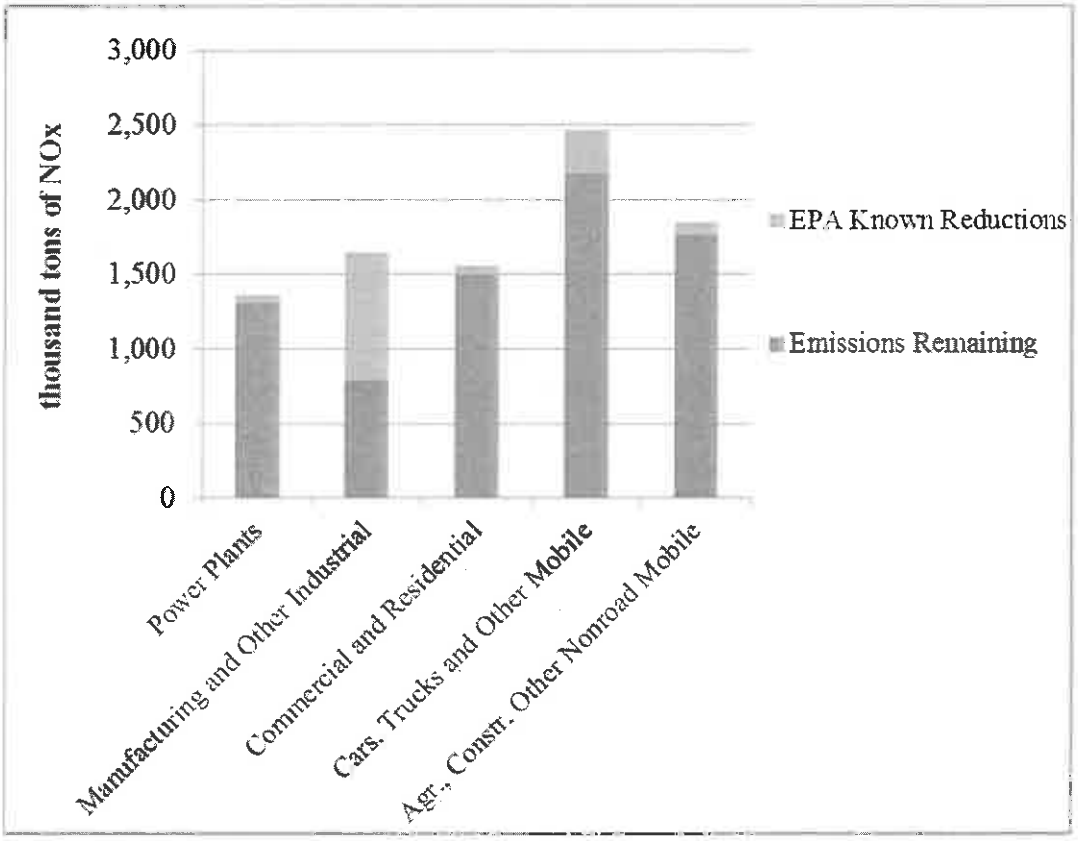
[18] The states without compliance costs would still have adverse economic impacts on net (despite competitive advantages relative to states with compliance costs) because of higher energy costs and reduced demand elsewhere in the country for their goods and services.

[19] The categories include two types of "point sources," which are large non-moving emitting equipment such as industrial boilers and electricity generating units (EGUs). The other three categories are "non-point sources," which means they are many small, diffuse sources. Of these "area sources" are non-moving equipment that are too individually small to be regulated as point sources are. Examples include commercial and residential water and space heaters as well as compressors along oil and natural gas pipelines. "Mobile sources" are small, diffuse and can be moved from place to place. Onroad mobile sources include cars and all sizes of trucks. Nonroad mobile sources include agricultural and construction equipment as well as transportation such as locomotives, airplanes, and boats.

Figure 7 shows the 2018 baseline emissions in states that will need to reduce NO_X emissions to meet a 60 ppb standard and the emission reductions due to EPA's "known" controls for the five emission categories. (The total 2018 baseline emission for these states across all five categories is 8.9 million tons). This information shows that most of the emissions that remain after EPA's "known" controls are from electricity generating units (EGUs) and non-point sources, while large industrial and manufacturing point sources are substantially controlled.

Figure 7: EPA Known NO_X Reductions from 2008-2010 Analysis and Remaining Emissions by General Categories of Emissions Sources in the 40 Non-Attaining States



Source: NERA calculations as explained in text

The list of known controls described in the previous section suggests that the "known" controls largely exhaust the options for retrofitting existing equipment with technology controls (e.g., installation of low-NO_X combustion devices and NO_X-destroying post-combustion devices). This explains why most of the known controls' effects are concentrated on the industrial and manufacturing emitters that comprise the "point source" category.[20] This evidence suggests that

[20] EPA's "known" controls for electric generation unit (EGU) sources (which are mostly from additional retrofits of selective catalytic reduction, SCR) have very little effect on EGU 2018 emissions. This is because almost all of the EGU point sources have already been retrofitted with NO_X controls in states projected to have nonattainment.

the bulk of the 2.6 million tons of "unknown" NO_X reductions will have to come various forms of capital stock replacement rather than further technology retrofits. While these replacements will likely include retirements of large coal-fired electricity generators, it also will likely become necessary to scrap and replace a wide array of very small sources, such as personal vehicles, individual pieces of construction equipment, and agricultural and landscaping equipment.

To indicate how EPA could develop more informed and evidence-based estimates of the costs of these remaining necessary types of NO_X reductions, we developed information on the costs of reducing emissions from two of the most significant categories of remaining NO_X emissions.

- *Retirement of coal-fired power plants.* If coal units are retired in states with large remaining NO_X reductions needs, and their generation is replaced by a cost-effective combination of natural gas and non-emitting generation, we estimate that an additional emissions reduction of about 1 million tons could be obtained. Our analyses indicate that these tons of reduction will cost an average of approximately $31,000/ton, but with costs ranging up to about $180,000/ton among the states. We replace the "known" power plant controls (retrofits) used in EPA's 2008-2010 analyses with these potential retirement controls in our analysis.

- *Scrapping of cars and light-duty trucks.* Cars and trucks will be much lower-emitting in 2018 than the fleet of vehicles on the road today, but in aggregate they account for a further potential reduction of 1 million tons, assuming every 2018 vehicle were to be scrapped in 2018 and replaced by either an electric vehicle (powered by natural gas generation) or a Tier 3 vehicle.[21] Using a model framework developed by an MIT researcher (Knittel 2009), we estimate the marginal cost per ton of reducing light-duty vehicle NO_X emissions by 10% through the early replacement of the highest-emitting cars and trucks would be in the range of $100,000/ton, a figure that escalates to about $500,000/ton to achieve about a 50% reduction.[22] Scrapping newer, lower-emitting cars

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Retirements rather than further retrofitting will be necessary to further reduce EGU emissions in these states and EPA did not consider retirements of equipment as a known control.

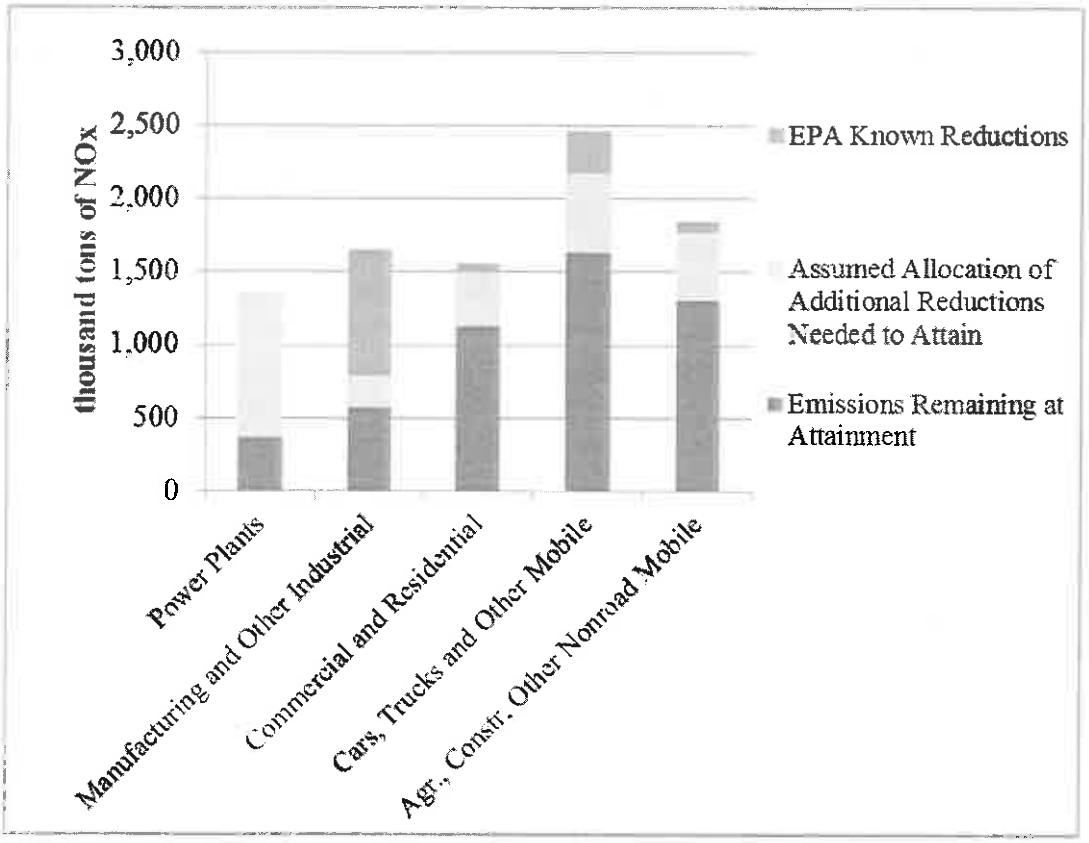
[21] The reduction is less than 1 million if one considers only vehicles in areas that contribute to nonattainment.

[22] We estimated this cost per ton removed for scrapping the marginal car that would achieve a 50% reduction of 2018 emissions from such vehicles following a methodology used in Knittel (2009) that is described in detail in Appendix C. Considering the subset of vehicle vintages making up 50% of light-duty vehicle emissions in 2020 (the first attainment year in our analysis), the newest vintage in the subset – or the marginal vehicles that would be scrapped in a program achieving 50% reduction – would have an average NO_X emission rate of 0.19 g/mile. Using an emission rate of 0.03 g/mile for replacement vehicles, along with an assumed average annual travel distance of 12,000 miles, the annual NO_X emission reduction of these marginal vehicles would be about 0.0022 tons per year for that vintage vehicle. We assumed the value of that vintage vehicle is $4,200, which, when annualized over its remaining useful vehicle life of about 4 years at a 5% discount rate, implies an annualized lost capital value of about $1,200 per year. Thus, the annualized cost per ton would be $1,200 / 0.0022 tons, or about $500,000 per ton. Sources and other information on these assumptions and calculations appear in Appendix C.

would cost more and generate fewer reductions per vehicle, so the incremental cost per ton rises as increasing percentages of the vehicle fleet are scrapped.

Replacing coal-fired EGUs would reduce NO_X emissions by about 1 million tons. Replacing *all* 2018 cars and light-duty vehicles would provide another 1 million tons of reduction. But other types of equipment certainly would become cost-effective to replace before one would go so far as to scrap all cars and light duty vehicles. We assume that the marginal cost-per-ton for these other sources rises similarly to the cost-per-ton we estimated for early turnover of different vintages of cars and light-duty trucks, as one indication of the potential costs that states would incur. Figure 8 shows the resulting mix of reductions assumed in our estimates of the compliance costs needed to achieve a 60 ppb ozone standard

Figure 8: NERA Analysis's Allocation of Additional Reductions Necessary to Attain a 60 ppb NAAQS to Categories of Emissions Sources in the 40 Non-Attaining States



Source: NERA calculations as explained in text

The dark green portions of the bars in Figure 8 shows EPA's "known" controls and the light green shows NERA's evidence-based assumptions regarding where "unknown" controls will likely come from. The remaining sum (shown in the blue bars) is now 5.0 million tons—the aggregate limit to achieve attainment for the states projected to be in nonattainment under

baseline 2018 emissions levels. NERA's estimates assume deep cuts in the EGU sector, where emissions are concentrated in a few sources and costs per ton are thus lower than for the many smaller sources among the non-point source categories (i.e., area, onroad mobile and nonroad mobile) NERA's assumptions on "unknown" controls outside of the EGU sector involve much smaller incremental percentage reductions than from EGUs; but because these will require programs such as scrapping vehicles and other small sources, they are expected to come at a substantially higher cost per ton than the EGU controls—even though we assume that the scrapping programs only target the oldest, highest-emitting of each type of NO_X-emitting equipment.

We developed estimates of EGU control costs using the results from N_{ew}ERA. In particular, using N_{ew}ERA, we modeled reduced generation from existing coal-fired power plants and replacement with natural gas power plants (or another energy source if optimal) as a potential NO_X emission reduction measure in each state where "known" non-EGU controls would be insufficient for 60 ppb compliance.[23] The potential costs and NO_X reductions from this measure thus are calculated within the N_{ew}ERA model.[24]

The next task related to our analysis of "unknown" costs was to use the information on the potential costs of phasing out NO_X-emitting mobile source emissions, in particular on the costs of scrapping older, high-emission rate passenger cars and light duty trucks and replacing them with new low-emission vehicles. Such vehicle scrapping programs were developed in California in the 1990's and applied most recently in 2009 (Car Allowance Rebate System) as part of the federal stimulus program. The cost per ton to scrap older vehicles varies with the age of the vehicles, since scrapping is both more expensive and generates fewer emission reductions for newer, lower emission vehicles. We used a framework developed by an MIT researcher who had evaluated the cost-effectiveness of vehicle scrappage to develop our empirical estimates. As noted above and explained in more detail in Appendix C, we used that framework to estimate that the marginal cost of scrapping sufficient passenger cars and light duty trucks to reduce 50% of their expected NO_X emissions would have a marginal cost of roughly $500,000 per ton. This same information indicates that the marginal cost per ton increases as the percent reduction in NO_X emissions increases, providing motivation and evidence for an increasing marginal cost curve.

[23] For convenience, in the report we sometimes refer to this option as "scrappage," although the specific assumption is that generation is not allowed (or is limited).

[24] As noted above, EPA (2014d, pp. ES-6 and ES-7) estimates that the proposed power sector CO_2 rule would reduce annual NO_X emissions by approximately 300,000 to 400,000 tons (depending on regulatory option, state or regional compliance approach, and measurement year). Our modeling of potential changes to coal-fired power plants for compliance with a new ozone NAAQS of 60 ppb would lead to a significantly larger NO_X reduction (as shown in Appendix C). Thus, the proposed power sector CO_2 rule would not change our conclusion that a new 60 ppb ozone NAAQS would have significant impacts on the power sector (and other sectors of the economy).

We used the information on vehicle scrapping to develop estimates of each state's marginal cost curve for "unknown" controls. In particular, as shown below, we assumed that the option of scrapping 50% of the vehicle fleet emissions would be undertaken at the part of each state's marginal cost curve corresponding to a 75% reduction from the baseline NO_X emission level. This anchor point, in combination with cost-per-ton information for "known" controls, determines our estimate of the slope of the "unknown" segment of each state's marginal cost curve. We estimated total annualized costs for "unknown" controls for each state using this slope and the necessary remaining tons of NO_X emission reductions after implementation of "known" controls.

Although the costs of "unknown" costs are highly uncertain and other data and assumptions could be used to derive state-by-state marginal cost curves, our calculations represent an effort to use evidence-based information—in particular, scrappage of existing coal-fired power plants and scrappage of existing vehicles of different vintages—to derive the marginal cost curve for the "unknown" controls. We emphasize that EPA needs to develop more specific information on control measures and their costs in order to reduce the enormous uncertainty in potential control costs due to the importance of "unknown" costs. Appendix C provides additional explanation and state-specific information based on these calculations.

c. NERA Marginal Cost Curve

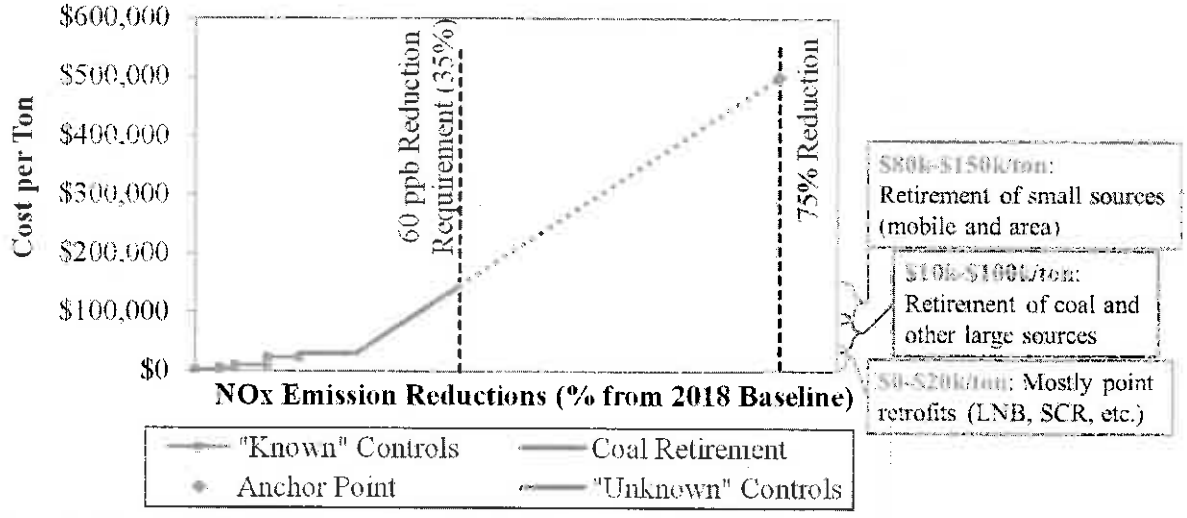
Figure 9 illustrates our methodology for constructing a state NO_X marginal cost curve including EPA "known" controls and an evidence-based approach to estimating costs of "unknown" controls. EPA "known" controls are shown in the lower left part of the curve. These controls generally are on point sources and are in a range up to about $20,000 per ton. The second segment of the curve reflects the costs of reduced generation from coal-fired power plants as estimated in N_{ew}ERA. State-level average costs of these generation controls range up to about $180,000 per ton; for purposes of developing the state-specific marginal cost curves, we only use generation controls in this segment of the cost curve if their costs are no greater than about $30,000 per ton; for more costly coal-fired generation controls, we incorporate the controls into the third segment of the curve.

The third segment of the curve reflects costs for additional "unknown" controls (beyond those related to retirement of coal-fired units). The evidence for this part of the curve is based upon the costs of scrapping existing motor vehicles. As discussed in Appendix C, we estimate that the marginal cost of scrapping 50 percent of NO_X emissions from light-duty motor vehicles is approximately $500,000 per ton. We also determined that the likely feasible emission reductions beyond the "known" controls would come from various non-point sources (including on-road sources such as cars and trucks, non-road sources such as trains, and area sources such as home and commercial heaters).

The slope for "unknown" controls in the illustrative marginal cost curve is based on two anchor points: the marginal cost per ton of reduced coal-fired generation (for the lower left point on the

curve) and the marginal cost per ton of vehicle scrappage at the 50 percent level as discussed above (for the upper right point on the curve). (For ease of exposition, we refer to reductions in coal-fired generation as coal scrappage in the figure.) As shown in the illustration, the segment for "unknown" controls required in a particular state is the solid line representing the additional controls that would be necessary (beyond EPA "known" controls and coal scrappage) for 60 ppb compliance. The dashed part of the curve illustrates additional "unknown" emission reductions for this state assumed to be "available," but not needed to achieve compliance. The estimated cost in each state is equal to the area under the solid segment of the curve, including "known" and "unknown" controls.

Figure 9: Illustration of a State NO_X Marginal Cost Curve Showing "Known" and "Unknown" Controls



Source: NERA illustration

The curve in Figure 9 provides a general illustration of our methodology for developing a marginal cost curve. As noted above, we use state-specific information on emissions and controls to develop our cost estimates. States differ substantially in their estimated marginal cost curve for several reasons. For one thing, in some states reduced coal-fired generation and replacement would cost more than $30,000 per ton, and in those cases we used the final EPA "known" control as the lower left end of the "unknown" control cost curve to estimate the slope. In addition, the share of controls represented by "unknown" controls required to achieve a 60 ppb ozone standard differs greatly among the states. Appendix C provides state-specific information related to the costs of reducing coal-fired generation and our estimates of "unknown" control costs.

A very large portion of our estimated costs of compliance at the national level is based on extrapolation beyond the list of the "known" control options that EPA prepared in its 2008-2010 analyses; as shown in Appendix C, the cost of "unknown" controls to achieve a 60 ppb ozone standard is about 60 times the cost of "known" controls (excluding reduced generation from existing coal-fired power plants). That such a large portion of the estimated compliance costs would be from controls that EPA has yet to identify highlights the dangers of implementing a policy for which most of the control options are unknown. As noted above, in its on-going NAAQS review, EPA should develop more complete information about additional controls to allow for a more reliable estimation of compliance costs than it produced in 2008-2010.

3. Step 3: Allocate Costs to N_{ew}ERA Sectors and Years

The third and final step in the development of the compliance cost inputs to N_{ew}ERA consisted of translating the annualized compliance costs into estimates of additional compliance costs in individual sectors and individual years.

a. Allocation to N_{ew}ERA Sectors

N_{ew}ERA models the economy using 10 sector aggregations (see Appendix A). Using EPA information from the 2008-2010 ozone review, including North American Industry Classification System (NAICS) codes for emission reductions from point sources, we matched the costs for "known" control measures to N_{ew}ERA sectors. The specific allocation of costs to sectors is based on state-level NO_X control information and baseline emissions and thus varies by state. Appendix D provides detailed allocation assumptions for each state.

For "unknown" control measures, we divided costs among four of the five EPA categories of emission sources (excluding the EGU category) based on the potential of each category for further emission reductions beyond the "known" controls.[25] Once associated with specific emission source categories, we matched costs for "unknown" controls to N_{ew}ERA sectors using the same logic and state-level calculations that were applied to the "known" control costs. Appendix D shows the cost information for "unknown" costs.

b. Allocation to N_{ew}ERA Modeling Years

The timing of costs in each state depends on when the state would be required to be in compliance with the 60 ppb standard as well as when different types of costs would be incurred. For states requiring NO_X emission reductions for 60 ppb compliance, we used EPA information on recent ozone monitor readings and the most recent EPA NO_X emission projections to develop

[25] We divided "unknown" control costs using the state-level shares of emissions remaining in the non-EGU point, area, onroad mobile, and nonroad mobile source categories after applying the "known" controls. We assumed that no additional EGU reductions are available after the reduced generation from coal-fired power plants, so we did not allocate any "unknown" control costs to the EGU emissions source category.

estimates of the compliance deadlines. The deadlines reflect estimated designations for each state as "Marginal," "Moderate," "Serious," "Severe," or "Extreme" nonattainment areas, using the system specified in the Clean Air Act and used by EPA for previous ozone standards. As noted above, some states would comply with 60 ppb under baseline conditions and would not incur any costs in our modeling. Note too that although EPA designates nonattainment areas as parts of states, we use a single designation and compliance deadline for each state. Figure 10 summarizes the state designations and compliance deadlines. The deadlines reflect the assumption that EPA would finalize the new ozone standard in 2015 and would finalize state designations in 2017.

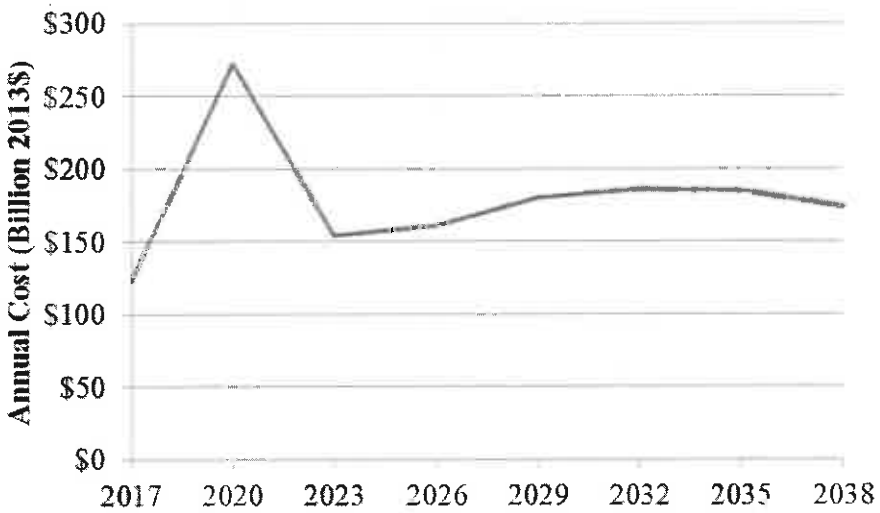
Figure 10: Classifications and Attainment Years for 60 ppb

Classification	Attainment Year	States
Compliant	N/A	8
Marginal	2020	5
Moderate	2023	32
Serious	2026	2
Severe-15	2032	0
Severe-17	2034	0
Extreme	2037	1

Notes: "N/A" denotes that attainment year is not applicable for compliant states.
State counts do not include Alaska, Hawaii, or the District of Columbia
Source: NERA calculations as explained in text

Based on these state designations and compliance deadlines, we developed estimates of the costs that would be incurred each year. We assumed that one-half of annualized costs represent capital costs that would be incurred before the compliance deadline and the other half represents operating costs that would be incurred each year from the compliance deadline onward. Figure 11 shows the resulting estimates of total potential compliance costs by year to achieve an ozone NAAQS of 60 ppb. The present value of the costs shown in this figure is $2.2 trillion (in 2013 dollars, calculated as of 2014, excluding costs related to coal-unit retirements (which are modeled endogenously in N_{ew}ERA). Appendix D provides details on the cost estimates. We ran the N_{ew}ERA model with these costs assigned to specific states, sectors, and years to estimate the economic impacts of a potential 60 ppb ozone NAAQS.

Figure 11: Potential U.S. Compliance Costs for 60 ppb by Year (Billion 2013$)



Notes: National summary figure reflects sum of state-specific modeling inputs.
Annual values are the average for that year and the following two years (*e.g.*, 2017 value is the average for the 2017-2019 period).
Compliance cost inputs in the figure do not include the cost of reduced coal-fired generation.
Source: NERA calculations as explained in text

C. Natural Gas Production Sensitivity Case

The methodology outlined above assumes that the U.S. natural gas extraction sector would be able to increase production without any constraints to meet increased natural gas demand associated with ozone NAAQS attainment actions. This assumption results in a large projected increase in U.S. natural gas production. However natural gas producers in areas that become nonattainment under a tighter ozone standard might face new requirements – such as the need to obtain air permits as well as emissions reduction credits ("offsets") for NO_X and/or VOCs – in order to develop new wells. Whether such permitting requirements will be applied to new oil and gas extraction nationally is a policy question that is in a state of flux at present; but some areas of the country already have these requirements and there are pressures for the EPA to make it a uniform requirement. Moreover, expansion of natural gas output will require additional gas processing facilities, which are already subject to the offsetting requirement if located in nonattainment areas. Obtaining offsets may be difficult and/or costly, particularly in more rural areas where there are few industrial emissions sources to create offset supply. If such barriers to continued new well development do emerge, the projected economic impacts of a 60 ppb ozone NAAQS could be substantially increased.

To consider the implications of possible constraints on energy production, we evaluated a natural gas production sensitivity case. This case was intended to provide an indication of the potential impacts if the 60 ppb standard effectively prevented additional natural gas production in

nonattainment areas. For this case we used the same attainment cost inputs as in the 60 ppb case, but we also assumed that total U.S. natural gas production would not increase beyond its 2020 level (from the 60 ppb scenario).[26] The motivation for this sensitivity case is the possibility that the majority of the natural gas producing regions will find themselves as part of a nonattainment area for 60 ppb and may face new air permit and emissions offset requirements in order to site new wells. Crude oil and natural gas activity has been linked to increases in ozone concentrations in several areas of the country (*e.g.*, Lyman and Shorthill 2013, Wyoming Outdoor Council *et al.* 2013, Travers 2013, and Colorado Department of Public Health and the Environment 2014). Jacus (2011) summarizes legal issues and cases related to crude oil/natural gas activity and air quality.

Thus for the sensitivity case, we assumed that natural gas production could not exceed the 2020 levels of production in the 60 ppb case. This case does not reflect the specific constraints that might result in individual states. Note that the sensitivity case implicitly assumes some new natural gas wells are developed after 2020. Indeed, if new wells were prohibited entirely, production levels would likely decline because of the decline over time in production from existing wells. Note also that this case excludes constraints on crude oil production, which also could be affected. Our sensitivity analysis highlights the need for EPA to evaluate potential impacts on domestic energy production in its forthcoming ozone RIA.

[26] Note that limits on natural gas production may also affect crude oil production, but we did not evaluate this.

III. STUDY RESULTS

This chapter summarizes the results of our analyses of the impacts of a 60 ppb ozone standard. The results are grouped into three major categories: (1) potential impacts on the overall U.S. economy and U.S. households; (2) potential impacts on the U.S. energy sectors; and (3) potential impacts on individual sectors and regions. As noted, we refer to our estimates as potential impacts because of the major uncertainties involved in the underlying estimates of compliance costs. The final section summarizes the major uncertainties in our results.

A. Potential Impacts on the U.S. Economy and U.S. Households

The potential effects of a 60 ppb ozone standard on the U.S. economy and U.S. households are substantial. This section presents estimated impacts of a 60 ppb standard on GDP, household consumption, the labor force, and total welfare.

1. Gross Domestic Product and Its Components

GDP is an economic measure of the entire economy. The components of GDP are consumption, investment, government spending, and net exports. Since the level of Federal government expenditures is assumed to remain constant, the changes in GDP are driven by changes in consumption, investment, and net exports. Figure 12 shows the estimated changes in GDP and its components due to the 60 ppb ozone standard. GDP declines from the baseline levels by an average of 1.2% per year during the period. Both consumption and investment decline as well.

Figure 12: Potential Impacts of 60 ppb Ozone Standard on U.S. Gross Domestic Product

	PV	2017	2020	2023	2026	2029	2032	2035	2038
GDP									
Baseline (Trillions)	$281	$18.3	$19.7	$21.2	$22.7	$24.4	$26.1	$27.8	$28.9
60 ppb Case (Trillions)	$278	$18.1	$19.5	$20.9	$22.4	$24.1	$25.7	$27.4	$28.6
% Change from Baseline	-1.2%	-0.8%	-1.2%	-1.4%	-1.3%	-1.3%	-1.3%	-1.2%	-1.2%
Consumption									
% Change from Baseline	-1.1%	-1.0%	-1.1%	-1.2%	-1.2%	-1.2%	-1.2%	-1.2%	-1.1%
Investment									
% Change from Baseline	-1.5%	0.5%	-1.7%	-2.4%	-1.9%	-1.5%	-1.6%	-1.7%	-1.8%
Net Exports									
% Change from Baseline	0.2%	0.5%	0.0%	-0.2%	0.3%	0.6%	0.4%	0.0%	-0.3%

Notes: Present value is from 2017 through 2040, discounted at 5% real discount rate. Government spending is also a component of GDP, but is unchanged from the baseline.
Source: NERA calculations as explained in text

2. Consumption per Household

One common economic metric of policy costs is the change in consumption per household (sometimes described as change in costs per household). It is important to note that, as with the other measures, the estimated change in consumption per household is a comprehensive figure that includes a large number of influences. This metric incorporates the financial gains due to increased demand for pollution control equipment. It also takes into account the many ways in which consumers and producers can change their behavior to limit financial losses from increases in prices due to the ozone standard. That is, this impact measure includes cost-minimizing adjustments to consumers' "market basket" of goods and services purchased and to their lifestyle/behavioral patterns. Similarly, the loss in consumption per household incorporates all the adjustments to inputs and production processes that businesses make to minimize the effects of compliance expenditures on the cost of their products or services. These adjustments can lead to non-financial losses and thus the change in consumption per household is not a complete measure of consumer losses. The full effects of the 60 ppb ozone standard include the qualitative effects of all such changes in personal choices and activities as well as the financial costs we report here.

Figure 13 shows the potential change in consumption per household over time. These results indicate that average potential household consumption would be reduced by about $1,190 in 2017 and by about $1,830 in 2038, with an average annual (present valued) reduction over the period from 2017 through 2040 of $1,570 per household.

Figure 13: Potential Impacts of 60 ppb Ozone Standard on U.S. Annual Change in Consumption per Household[27]

	Avg.	2017	2020	2023	2026	2029	2032	2035	2038
Change in Average Consumption per Household	-$1,570	-$1,190	-$1,430	-$1,590	-$1,640	-$1,760	-$1,830	-$1,850	-$1,830

Note: Average is the levelized average over 2017-2040, annualized using a 5% real discount rate.
Source: NERA calculations as explained in text

[27] PV is a levelized value over the 2017 through 2040 time period.

3. Labor Market

Figure 14 focuses on several dimensions of projected impacts on income from labor ("worker income") as a result of the 60 ppb ozone standard. The ozone standard on balance would lower potential wage rates by an average of 1.2% over the period from 2017 through 2040. Wage rates decline because companies have higher costs and lower labor productivity due to compliance costs. Lower real wage rates reduce workers' incomes even if they continue to work the same number of hours. However, a lower real wage rate also decreases people's desire to work. With fewer hours worked, total labor income declines by a greater percentage than does the wage rate (an average of 1.9% over the period). These are the net equilibrium effects on labor in the aggregate, and include the positive benefits of increased labor demand in sectors providing pollution control equipment and technologies.

Figure 14: Potential Impacts of 60 ppb Ozone Standard on Labor

	Avg.	2017	2020	2023	2026	2029	2032	2035	2038
Baseline Job-Equivalents (millions)	155.7	145.2	148.4	150.3	153.5	157.2	160.6	163.7	167.0
60 ppb Case									
Real Wage Rate (% Change from Baseline)	-1.2%	-0.8%	-2.0%	-1.3%	-1.2%	-1.3%	-1.2%	-1.2%	-1.1%
Change in Labor Income (% Change from Baseline)	-1.9%	-0.7%	-2.2%	-2.0%	-2.0%	-2.1%	-2.1%	-2.1%	-1.9%
Job-Equivalents (Change from Baseline, millions)	-2.9	-0.9	-3.2	-3.0	-3.0	-3.3	-3.4	-3.3	-3.2

Notes: Average is the simple average over 2017-2040. Total job-equivalents equals total labor income change divided by the average annual income per job. This does not represent a projection of numbers of workers that may need to change jobs and/or be unemployed, as some or all of it could be spread across workers who remain employed.

Source: **NERA calculations as explained in text**

The total reduction in potential labor income is spread over many workers, most of whom may continue to work, but the dollar magnitude of the reduction in labor income can be placed in context by estimating the equivalent number of average jobs that such labor payments would fund under baseline wage rates. To state the potential labor income changes in terms of such "job-equivalents," we divide the change in labor income by the annual baseline income for the average job (see Figure 14). A loss of one job-equivalent does not necessarily mean one fewer employed person—it may be manifested as a combination of fewer people working and less income per worker. However, this measure allows us to express employment-related impacts in

terms of an equivalent number of employees earning the average prevailing wage.[28] Note that the NewERA model, like many other similar economic models, does not develop projections of unemployment rates or layoffs associated with reductions in labor income; modeling such largely transitional phenomena requires a different type of modeling methodology; our methodology considers only the long-run, equilibrium impact levels.

The projected impacts of a 60 ppb ozone standard on potential labor income are substantial. Potential labor income declines by about 0.7% to 2.2% throughout the period, resulting in potential annual job-equivalent losses that average about 2.9 million job-equivalents.

4. Economic Welfare

Economic welfare is a concept used by economists that relates to the overall utility that individuals experience from the economy. In NewERA, welfare is measured by the sum of the values of household consumption and leisure. The potential effects of the 60 ppb ozone standard lead to a potential average U.S. welfare loss over the entire modeling horizon of 0.90%, expressed as percentage changes relative to the baseline, over the time period of our study.

B. Potential Impacts on U.S. Energy System

The transformations required to meet a 60 ppb ozone standard could have substantial impacts on the U.S. energy system. These potential impacts include effects on fossil fuel markets and electricity prices.

1. Fossil Fuel Markets

We estimated controls to achieve a 60 ppb ozone standard include elimination of generation from coal-fired power plants in certain nonattainment states. These changes would lead to higher costs for consuming other fossil fuels and reduced production and consumption of coal in the long term. Figure 15 shows the potential impacts on coal and natural gas production due to the 60 ppb ozone standard. The significant potential declines in steam coal consumption reflect the reduced generation from coal-fired generators. Potential natural gas consumption increases substantially due to fuel switching from coal to natural gas in the electricity sector.[29]

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[28] Such a "job-equivalent" estimate is comparable to determining the minimum number of workers that would lose their entire income (their "job") if the full brunt of the regulation were concentrated on the smallest number of workers.

[29] Natural gas production decreases slightly in 2020 (before the retirement of coal units and the resulting increase in natural gas generation) due to reduced overall economic activity.

Figure 15: Potential Impacts of a 60 ppb Ozone Standard on Fossil Fuel Production

	2017	2020	2023	2026	2029	2032	2035	2038
Steam Coal (Quadrillion Btu)								
Baseline	17.0	17.0	18.0	18.6	18.9	19.1	19.4	19.4
60 ppb Case	15.4	14.8	8.2	6.0	6.0	6.0	6.0	6.0
Change	-1.6	-2.2	-9.8	-12.7	-12.9	-13.2	-13.4	-13.4
Natural Gas (Quadrillion Btu)								
Baseline	26.7	29.4	31.3	32.7	33.6	34.6	35.7	36.7
60 ppb Case	26.8	28.9	34.6	37.4	38.1	39.0	40.2	41.2
Change	0.17	-0.6	3.2	4.7	4.4	4.4	4.5	4.5

Source: NERA calculations as explained in text

Figure 16 shows the potential impacts on the prices of fossil fuels. The average Henry Hub natural gas prices would increase by an average of almost 10% over the period from 2017 through 2040. Delivered natural gas prices to residential and industrial customers potentially would increase on average by 7% and 12%, respectively, over the same time period. Note that part of the increase in delivered natural gas prices reflects the increase in pipeline transportation costs due to control costs for reductions in NO_X emissions in the pipeline system that would be recovered through tariff rates.

2. Electricity Sector

Figure 17 shows the residential and industrial sector delivered electricity prices in the baseline and under the potential impacts of the 60 ppb ozone standard. In the baseline, both residential and industrial electricity prices are projected to increase primarily due to increasing fuel prices over time. The 60 ppb ozone standard is projected to lead to a potential increase in average delivered residential electricity price of 3.3% over the period from 2017 through 2040. Average delivered industrial electricity prices are projected to increase potentially by 5.5% over the same period.

Figure 18 shows projected potential physical impacts on the electricity sector in terms of coal electricity unit capacity and overall electricity demand. Projected reductions in coal-fired power plant generation to reduce NO_X emissions in certain nonattainment states would lead to cost increases from the need to use more expensive sources of electricity. This would result in reductions in potential electricity demand shown in the table below. The average reduction is 3.1% over the period from 2017 through 2040.

Figure 16: Potential Impacts of a 60 ppb Ozone Standard on Fossil Fuel Commodity Prices ($/MMBtu for Natural Gas, $/gallon for Gasoline)

	Avg.	2017	2020	2023	2026	2029	2032	2035	2038
Baseline Prices ($/MMBtu for natural gas and $/gallon for gasoline)									
Henry Hub Natural Gas	$6.02	$4.42	$4.87	$5.25	$5.70	$6.18	$6.68	$7.29	$7.76
Natural Gas Delivered (Residential)	$13.77	$12.25	$12.56	$12.96	$13.46	$14.00	$14.51	$14.99	$15.43
Natural Gas Delivered (Industrial)	$8.43	$6.80	$7.15	$7.58	$8.11	$8.67	$9.21	$9.73	$10.19
Gasoline	$3.56	$3.19	$3.25	$3.40	$3.50	$3.59	$3.70	$3.86	$4.00
60 ppb Case ($/MMBtu for natural gas and $/gallon for gasoline)									
Henry Hub Natural Gas	$6.65	$4.47	$4.73	$6.02	$6.73	$7.05	$7.50	$8.12	$8.57
Natural Gas Delivered (Residential)	$14.79	$12.63	$12.74	$14.15	$14.96	$15.36	$15.82	$16.19	$16.45
Natural Gas Delivered (Industrial)	$9.49	$7.22	$7.39	$8.82	$9.66	$10.08	$10.55	$10.96	$11.24
Gasoline	$3.57	$3.20	$3.25	$3.41	$3.52	$3.61	$3.72	$3.87	$4.01
60 ppb Case (% Increase from Baseline)									
Henry Hub Natural Gas	9.9%	1.1%	-2.7%	15%	18%	14%	12%	11%	10%
Natural Gas Delivered (Residential)	7.3%	3.1%	1.4%	9.1%	11%	9.8%	9.0%	8.0%	6.6%
Natural Gas Delivered (Industrial)	12%	6.2%	3.3%	16%	19%	16%	15%	13%	10%
Gasoline	0.4%	0.3%	0.0%	0.5%	0.5%	0.5%	0.5%	0.3%	0.2%

Note: Average is the simple average over 2017-2040.
Source: NERA calculations as explained in text

Figure 17: Potential Impacts of a 60 ppb Ozone Standard on Delivered Electricity Prices (¢/kWh)

	Avg.	2017	2020	2023	2026	2029	2032	2035	2038
Baseline Prices (¢/kWh)									
Residential	14.5¢	12.9¢	13.3¢	14.0¢	14.4¢	14.8¢	15.3¢	15.6¢	15.4¢
Industrial	9.4¢	7.8¢	8.1¢	8.9¢	9.3¢	9.8¢	10.3¢	10.7¢	10.5¢
60 ppb Case (¢/kWh)									
Residential	14.9¢	13.2¢	13.7¢	14.6¢	15.1¢	15.4¢	15.8¢	15.9¢	15.8¢
Industrial	9.9¢	8.1¢	8.6¢	9.5¢	10.1¢	10.4¢	10.8¢	11.0¢	10.9¢
60 ppb Case (% Increase from Baseline)									
Residential	3.3%	2.3%	3.2%	4.4%	5.3%	4.1%	3.0%	1.8%	2.4%
Industrial	5.5%	4.2%	6.0%	7.5%	8.6%	6.7%	4.6%	2.9%	3.9%

Note: Average is the simple average over 2017-2040.
Source: NERA calculations as explained in text

Figure 18: Potential Impacts of a 60 ppb Ozone Standard on Electricity Sector

	2017	2020	2023	2026	2029	2032	2035	2038
Baseline								
Coal-Fired Capacity (GW)	262	259	257	257	257	257	257	257
Electricity Demand (TWh)	4,150	4,250	4,370	4,480	4,580	4,650	4,750	4,850
60 ppb Case								
Coal-Fired Capacity (GW)	229	221	166	160	160	158	158	156
Electricity Demand (TWh)	4,080	4,140	4,210	4,290	4,400	4,500	4,620	4,710
60 ppb Case (Change from Baseline)								
Coal-Fired Capacity (GW)	-34	-38	-92	-97	-97	-99	-99	-101
Electricity Demand (%)	-1.7%	-2.7%	-3.7%	-4.2%	-3.9%	-3.2%	-2.6%	-3.0%

Note: Average is the simple average over 2017-2040.
Source: NERA calculations as explained in text

C. Potential Impacts on U.S. Sectors and Regions

Although all sectors of the U.S. economy would be affected by a 60 ppb ozone standard, both directly through increased emissions control costs and indirectly through the overall impact on the economy, there are noticeable differences across sectors.

1. Potential Sectoral Impacts

Figure 19 shows the estimated potential changes in sectoral output for 10 sectors. The reduction in coal output and increase in natural gas output are largely results of the scrappage of coal-fired power plants in some nonattainment areas and the resulting shift toward natural gas generation. While agriculture and commercial transportation have the largest percentage reductions in output among the non-energy sectors, the largest absolute reductions over the period from 2017 through 2040 are in the commercial/services and manufacturing sectors since these are much larger sectors.

Figure 19: Potential Impacts of a 60 ppb Ozone Standard on Sectoral Output (Percentage Change from Baseline)

	Avg.	2017	2020	2023	2026	2029	2032	2035	2038
Non-Energy Sectors									
Agriculture	-2.2%	-0.1%	-2.7%	-2.6%	-2.3%	-2.6%	-2.7%	-2.6%	-2.3%
Commercial/Services	-0.9%	-0.4%	-1.1%	-1.0%	-0.9%	-1.0%	-1.1%	-1.0%	-1.0%
Manufacturing	-0.6%	1.7%	-0.9%	-0.9%	-0.7%	-0.9%	-1.2%	-1.2%	-1.1%
Commercial Transportation	-1.9%	-0.8%	-2.2%	-1.9%	-2.0%	-2.1%	-2.1%	-1.9%	-1.8%
Commercial Trucking	-1.1%	-0.1%	-1.4%	-1.2%	-1.1%	-1.2%	-1.3%	-1.2%	-1.2%
Energy Sectors									
Coal	-52%	-8.3%	-12%	-55%	-67%	-67%	-68%	-68%	-69%
Natural Gas	9.2%	0.6%	-2.0%	10%	14%	13%	13%	12%	12%
Refining	-1.8%	-0.5%	-1.5%	-1.9%	-2.0%	-2.3%	-2.4%	-2.1%	-2.0%
Crude Oil	-0.1%	0.4%	0.0%	0.1%	0.0%	-0.1%	-0.3%	-0.3%	-0.3%
Electricity	-3.1%	-1.7%	-2.6%	-3.6%	-4.2%	-3.9%	-3.2%	-2.6%	-2.9%

Note: Average is the simple average over 2017-2040.
Source: NERA calculations as explained in text

2. Potential Regional Impacts

The potential impacts of a 60 ppb ozone standard vary by region. Regions fare better or worse than the U.S. average primarily due to each region's attainment costs and sectoral output mix.

Figure 20 shows the estimated potential changes in gross regional product for the eleven regions in the N_{ew}ERA model. California, the Mid-Atlantic and Upper Midwest regions have the largest percentage impacts, though all regions experience some reduction in gross regional product. As noted above, total U.S. gross domestic product decreases by about 1.2% per year from baseline during the model period.

Figure 20: Potential Impacts of a 60 ppb Ozone Standard on Gross Regional Product (Percentage Change Relative to Baseline)

Region	PV	2017	2020	2023	2026	2029	2032	2035	2038
Arizona and Mountain States	-0.8%	-0.7%	-0.9%	-1.0%	-0.8%	-0.9%	-0.7%	-0.7%	-0.6%
California	-2.2%	-1.3%	-1.5%	-1.7%	-2.4%	-3.0%	-3.2%	-2.9%	-2.7%
Florida	-0.4%	-0.7%	-0.5%	-0.5%	-0.2%	0.1%	-0.1%	-0.3%	-1.0%
Mid-America	-0.6%	0.3%	-0.4%	-0.5%	-0.9%	-0.8%	-0.8%	-0.8%	-0.8%
Mid-Atlantic	-1.6%	-1.7%	-2.1%	-2.4%	-1.5%	-1.3%	-1.3%	-1.0%	-1.0%
Mississippi Valley	-1.3%	-1.1%	-1.4%	-1.6%	-1.3%	-1.3%	-1.3%	-1.1%	-1.0%
New York/New England	-1.4%	-1.5%	-1.6%	-1.9%	-1.3%	-1.3%	-1.2%	-1.2%	-1.1%
Pacific Northwest	-0.6%	-0.6%	-0.5%	-0.6%	-0.5%	-0.9%	-0.5%	-0.5%	-0.3%
Southeast	-0.9%	-0.7%	-1.1%	-0.9%	-1.0%	-1.1%	-1.0%	-1.0%	-0.9%
Texas, Oklahoma, Louisiana	-0.3%	0.0%	-0.6%	-0.6%	-0.1%	0.0%	-0.1%	-0.6%	-0.5%
Upper Midwest	-1.6%	0.4%	-1.3%	-1.9%	-3.2%	-2.2%	-2.0%	-1.8%	-1.9%
U.S.	**-1.2%**	**-0.8%**	**-1.2%**	**-1.4%**	**-1.3%**	**-1.3%**	**-1.3%**	**-1.2%**	**-1.2%**

Note: Present value is from 2017 through 2040, discounted at 5% real discount rate.
Source: NERA calculations as explained in text

Figure 21 shows projected potential changes in average consumption per household by region over the period from 2017 through 2040. The regional patterns are similar to gross regional product, and all regions experience a decrease in average consumption per household.

Figure 22 shows potential changes in job-equivalents by region (relative to baseline) due to the 60 ppb ozone standard. All regions experience a decrease in job-equivalents, though potential impacts vary considerably by region.

Figure 21: Potential Impacts of a 60 ppb Ozone Standard on Regional Consumption per Household – Change in Consumption per Household Relative to Baseline ($/HH)

Region	PV	2017	2020	2023	2026	2029	2032	2035	2038
Arizona and Mountain States	-$690	-$550	-$610	-$680	-$740	-$780	-$800	-$800	-$780
California	-$2,910	-$1,710	-$2,130	-$2,600	-$3,170	-$3,710	-$4,060	-$4,160	-$4,100
Florida	-$450	-$340	-$410	-$450	-$510	-$530	-$520	-$510	-$480
Mid-America	-$850	-$620	-$670	-$770	-$890	-$990	-$1,080	-$1,140	-$1,160
Mid-Atlantic	-$2,520	-$2,110	-$2,410	-$2,670	-$2,510	-$2,660	-$2,740	-$2,820	-$2,840
Mississippi Valley	-$1,550	-$1,230	-$1,450	-$1,610	-$1,590	-$1,690	-$1,750	-$1,780	-$1,790
New York/New England	-$2,490	-$1,870	-$2,380	-$2,610	-$2,550	-$2,680	-$2,800	-$2,890	-$2,950
Pacific Northwest	-$730	-$590	-$650	-$680	-$760	-$820	-$880	-$900	-$890
Southeast	-$1,060	-$880	-$1,020	-$1,070	-$1,080	-$1,150	-$1,190	-$1,200	-$1,180
Texas, Oklahoma, Louisiana	-$1,070	-$770	-$1,010	-$1,190	-$1,240	-$1,240	-$1,180	-$1,120	-$1,080
Upper Midwest	-$1,770	-$1,440	-$1,720	-$1,850	-$1,800	-$1,860	-$1,930	-$1,980	-$2,000
U.S.	**-$1,570**	**-$1,190**	**-$1,430**	**-$1,590**	**-$1,640**	**-$1,760**	**-$1,830**	**-$1,850**	**-$1,830**

Note: Present value is from 2017 through 2040, discounted at 5% real discount rate.
Source: NERA calculations as explained in text

Figure 22: Potential Impacts of a 60 ppb Ozone Standard on Regional Job-Equivalents - Change in Job-Equivalents Relative to Baseline (thousands)

Region	Avg.	2017	2020	2023	2026	2029	2032	2035	2038
Arizona and Mountain States	-86	-12	-96	-96	-88	-96	-100	-101	-98
California	-608	-123	-319	-408	-529	-762	-899	-926	-895
Florida	-53	-11	-47	-63	-57	-61	-66	-58	-57
Mid-America	-69	8	-62	-72	-68	-81	-90	-96	-87
Mid-Atlantic	-364	-175	-541	-387	-381	-367	-372	-351	-337
Mississippi Valley	-289	-94	-350	-329	-314	-319	-319	-302	-286
New York/New England	-333	-160	-468	-366	-346	-347	-333	-330	-312
Pacific Northwest	-54	-12	-49	-56	-49	-59	-68	-70	-65
Southeast	-303	-101	-378	-328	-302	-326	-343	-332	-311
Texas, Oklahoma, Louisiana	-313	-83	-319	-358	-385	-391	-344	-310	-314
Upper Midwest	-453	-136	-581	-539	-485	-486	-487	-467	-445
U.S.	**-2,920**	**-900**	**-3,210**	**-3,000**	**-3,000**	**-3,290**	**-3,420**	**-3,340**	**-3,210**

Note: Average is the simple average over 2017-2040.
Source: NERA calculations as explained in text

Figure 23 shows potential changes in economic welfare by region (relative to baseline) due to the 60 ppb ozone standard. All regions are projected to suffer negative economic outcomes, but some regions fare better or worse than the U.S. average depending on the quantity and cost of NO_X reduction required for compliance, along with their natural resource base (regions with natural gas have some benefits associated with increased production that offsets some of their compliance costs, while coal producing regions have negative consequences from reduced coal production on top of their compliance costs).

Figure 23: Potential Impacts of a 60 ppb Ozone Standard on Regional Welfare (Percentage Change Relative to Baseline)

Region	PV
Arizona and Mountain States	-0.44%
California	-1.41%
Florida	-0.34%
Mid-America	-0.46%
Mid-Atlantic	-1.28%
Mississippi Valley	-0.86%
New York/New England	-1.11%
Pacific Northwest	-0.47%
Southeast	-0.70%
Texas, Oklahoma, Louisiana	-0.64%
Upper Midwest	-1.12%
U.S.	**-0.90%**

Note: Present value is from 2017 through 2040, discounted at 5% real discount rate
Source: NERA calculations as explained in text

D. Sensitivity Case with Limits on Natural Gas Production

For the sensitivity case we used the same attainment cost inputs as in the 60 ppb case, but we also assumed that total U.S. natural gas production would not increase beyond its 2020 level (from the 60 ppb scenario). Figure 24 shows natural gas production under baseline conditions, with a 60 ppb ozone standard but no restrictions on natural gas production ("60 ppb"), and with natural gas production limited to 2020 production in the 60 ppb case. Natural gas production is 28.9 quads in 2020 in the 60 ppb case (without any natural gas constraints), so we limited natural gas production to 28.9 quads after 2020 in the sensitivity case.

Figure 24: Potential Impacts of a 60 ppb Ozone Standard on Natural Gas Production (Quadrillion Btu)

	2017	2020	2023	2026	2029	2032	2035	2038
Baseline	26.7	29.4	31.3	32.7	33.6	34.6	35.7	36.7
60 ppb	26.8	28.9	34.6	37.4	38.1	39.0	40.2	41.2
Production Sensitivity	26.8	28.9	28.9	28.9	28.9	28.9	28.9	28.9

Source: NERA calculations as explained in text

1. Potential Impacts on the U.S. Economy and U.S. Households

Figure 25 shows the estimated potential changes in GDP and its components due to the 60 ppb ozone standard with assumed limits on natural gas production. GDP declines from the baseline levels by approximately 1.6% per year during the period, a potential reduction roughly 30% higher than in the 60 ppb case without limits on natural gas production. Both consumption and investment decline as well.

Figure 25: Potential Impacts of a 60 ppb Ozone Standard on U.S. Gross Domestic Product and Components (Sensitivity Case)

	PV	2017	2020	2023	2026	2029	2032	2035	2038
GDP									
Baseline (Trillions)	$281	$18.3	$19.7	$21.2	$22.7	$24.4	$26.1	$27.8	$28.9
Sensitivity (Trillions)	$277	$18.1	$19.5	$20.8	$22.3	$23.9	$25.6	$27.3	$28.4
% Change from Baseline	-1.6%	-0.9%	-1.3%	-1.6%	-1.8%	-1.9%	-1.9%	-1.9%	-1.8%
Consumption									
% Change from Baseline	-1.5%	-1.1%	-1.4%	-1.5%	-1.6%	-1.7%	-1.7%	-1.6%	-1.6%
Investment									
% Change from Baseline	-2.5%	0.0%	-2.1%	-3.1%	-2.7%	-2.8%	-3.1%	-3.4%	-3.5%
Net Exports									
% Change from Baseline	-1.0%	2.6%	-0.7%	-1.3%	-1.2%	-0.3%	0.2%	0.0%	-0.6%

Note: Present value is from 2017 through 2040, discounted at 5% real discount rate. Government spending is also a component of GDP, but is unchanged from the baseline.
Source: NERA calculations as explained in text

Figure 26 shows the potential change in consumption per household over time in the sensitivity case. These results indicate that average household consumption would be reduced by about $1,370 in 2017 and by about $2,580 in 2038, with an average annual (present valued) reduction over the period from 2017 through 2040 of $2,040 per household.

Figure 26: Potential Impacts of a 60 ppb Ozone Standard on U.S. Consumption per Household (Sensitivity Case)

	Avg.	2017	2020	2023	2026	2029	2032	2035	2038
Change in Average Consumption per Household	-$2,040	-$1,370	-$1,710	-$2,000	-$2,210	-$2,430	-$2,550	-$2,590	-$2,580

Note: Annualized average is from 2017 through 2040, discounted at 5% real discount rate.
Source: NERA calculations as explained in text

Figure 27 shows projected impacts on income from labor ("worker income") as a result of the 60 ppb ozone standard with assumed limits on natural gas production. The ozone standard sensitivity on balance would lower wage rates by an average of 2.0% over the period from 2017 through 2040. Wage rates decline because companies have higher costs and lower labor productivity due to compliance costs. Lower real wage rates reduce workers' incomes even if they continue to work the same number of hours. However, a lower real wage rate also decreases people's desire to work. With fewer hours worked, total labor income declines by a greater percentage than does the wage rate (an average of 2.7% over the period). These are the net equilibrium effects on labor in the aggregate, and include the potential positive benefits of increased labor demand in sectors providing pollution control equipment and technologies.

In the sensitivity case, the projected potential impacts of a 60 ppb ozone on labor income are substantial, particularly in the later years. Labor income declines by about 0.7% to 3.3% throughout the period, resulting in job-equivalent losses that average about 4.3 million job-equivalents (compared to 2.9 million average annual job-equivalent losses in the basic 60 ppb case).

The effects of the 60 ppb ozone standard with assumed limits on natural gas production lead to an average potential U.S. welfare loss over the entire modeling horizon of 1.12%, expressed as a percentage change relative to the baseline, over the time period of our study.

Figure 27: Potential Impacts of a 60 ppb Ozone Standard on Labor (Sensitivity Case)

	Avg.	2017	2020	2023	2026	2029	2032	2035	2038
Baseline Job-Equivalents (millions)	155.7	145.2	148.4	150.3	153.5	157.2	160.6	163.7	167.0
Sensitivity Case									
Real Wage Rate (% Change from Baseline)	-2.0%	-0.9%	-2.1%	-1.9%	-2.2%	-2.3%	-2.3%	-2.2%	-2.1%
Change in Labor Income (% Change from Baseline)	-2.7%	-0.7%	-2.3%	-2.7%	-3.1%	-3.3%	-3.3%	-3.3%	-3.1%
Job-Equivalents (Change from Baseline, millions)	-4.3	-1.1	-3.4	-4.1	-4.8	-5.1	-5.3	-5.4	-5.2

Note: Average is the simple average over 2017-2040. Total job-equivalents equals total labor income change divided by the average annual income per job. This does not represent a projection of numbers of workers that may need to change jobs and/or be unemployed, as some or all of it could be spread across workers who remain employed.

Source: NERA calculations as explained in text

2. Potential Impacts on U.S. Energy System

Figure 28 shows the potential impacts on coal and natural gas production due to the 60 ppb ozone standard with assumed limits on natural gas production. The significant declines in coal production reflect the reductions in coal-fired generation. Natural gas production is restricted by assumption in the sensitivity case.

Figure 28: Potential Impacts of a 60 ppb Ozone Standard on Fossil Fuel Production (Sensitivity Case)

	2017	2020	2023	2026	2029	2032	2035	2038
Steam Coal (Quadrillion Btu)								
Baseline	17.0	17.0	18.0	18.6	18.9	19.1	19.4	19.4
Sensitivity Case	15.3	14.0	8.3	5.9	6.0	6.0	6.0	6.0
Change	-1.7	-3.0	-9.7	-12.7	-12.9	-13.1	-13.4	-13.4
Natural Gas (Quadrillion Btu)								
Baseline	26.7	29.4	31.3	32.7	33.6	34.6	35.7	36.7
Sensitivity Case	26.8	28.9	28.9	28.9	28.9	28.9	28.9	28.9
Change	0.2	-0.5	-2.4	-3.8	-4.6	-5.7	-6.8	-7.8

Source: NERA calculations as explained in text

Figure 29 shows the potential impacts on the prices of fossil fuels. Due to assumed limits on the production of natural gas in the sensitivity case, Henry Hub natural gas prices would increase by an average of 66% over the period from 2017 through 2040 (compared to about 10% in the 60 ppb case, shown in Figure 16). Delivered natural gas prices to residential and industrial customers potentially would increase on average by 32% and 52%, respectively, over the same time period. Note that part of the increase in delivered natural gas prices reflects the increase in pipeline transportation costs due to control costs for reductions in NO_X emissions in the pipeline system that would be recovered through tariff rates.

Figure 29: Potential Impacts of a 60 ppb Ozone Standard on Fossil Fuel Commodity Prices (Sensitivity Case)

	Avg.	2017	2020	2023	2026	2029	2032	2035	2038
Baseline Prices ($/MMBtu for natural gas and $/gallon for gasoline)									
Henry Hub Natural Gas	$6.02	$4.42	$4.87	$5.25	$5.70	$6.18	$6.68	$7.29	$7.76
Natural Gas Delivered (Residential)	$13.77	$12.25	$12.56	$12.96	$13.46	$14.00	$14.51	$14.99	$15.43
Natural Gas Delivered (Industrial)	$8.43	$6.80	$7.15	$7.58	$8.11	$8.67	$9.21	$9.73	$10.19
Gasoline	$3.56	$3.19	$3.25	$3.40	$3.50	$3.59	$3.70	$3.86	$4.00
Sensitivity Case ($/MMBtu for natural gas and $/gallon for gasoline)									
Henry Hub Natural Gas	$9.97	$4.45	$4.36	$9.10	$12.09	$12.30	$12.72	$13.25	$13.78
Natural Gas Delivered (Residential)	$18.16	$12.62	$12.38	$17.03	$20.09	$20.50	$20.99	$21.24	$21.54
Natural Gas Delivered (Industrial)	$12.79	$7.21	$7.03	$11.73	$14.83	$15.24	$15.75	$16.03	$16.34
Gasoline	$3.60	$3.21	$3.26	$3.45	$3.56	$3.65	$3.76	$3.91	$4.05
Sensitivity Case (% Increase from Baseline)									
Henry Hub Natural Gas	66%	0.8%	-10%	73%	112%	99%	90%	82%	78%
Natural Gas Delivered (Residential)	32%	3.0%	-1.4%	31%	49%	46%	45%	42%	40%
Natural Gas Delivered (Industrial)	52%	6.0%	-1.6%	55%	83%	76%	71%	65%	60%
Gasoline	1.3%	0.7%	0.3%	1.4%	1.8%	1.7%	1.7%	1.4%	1.2%

Note: Average is the simple average over 2017-2040.
Source: NERA calculations as explained in text

Figure 30 shows the residential and industrial sector delivered electricity prices in the baseline and under the potential impacts of the 60 ppb ozone standard with assumed limits on natural gas production. In the baseline, both residential and industrial electricity prices are projected to increase primarily due to increasing fuel prices over time. The 60 ppb ozone standard with limits on natural gas production is projected to lead to a potential increase in average delivered residential electricity price of 15% over the period from 2017 through 2040. Average delivered industrial electricity prices are projected to increase by 23% over the same period.

Figure 30: Potential Impacts of a 60 ppb Ozone Standard on Delivered Electricity Prices (¢/kWh)

	Avg.	2017	2020	2023	2026	2029	2032	2035	2038
Baseline Prices (¢/kWh)									
Residential	14.5¢	12.9¢	13.3¢	14.0¢	14.4¢	14.8¢	15.3¢	15.6¢	15.4¢
Industrial	9.4¢	7.8¢	8.1¢	8.9¢	9.3¢	9.8¢	10.3¢	10.7¢	10.5¢
Sensitivity Case (¢/kWh)									
Residential	16.6¢	13.3¢	13.9¢	16.1¢	17.4¢	17.6¢	18.0¢	18.5¢	18.4¢
Industrial	11.6¢	8.2¢	8.7¢	11.1¢	12.4¢	12.7¢	13.1¢	13.6¢	13.5¢
Sensitivity Case (% Increase from Baseline)									
Residential	15%	3.0%	4.7%	15%	21%	19%	18%	18%	19%
Industrial	23%	5.2%	8.0%	25%	33%	29%	27%	27%	29%

Note: Average is the simple average over 2017-2040.
Source: NERA calculations as explained in text

Figure 31 shows projected potential physical impacts on the electricity sector in terms of coal electricity unit retirements and overall electricity demand in the sensitivity case. Projected potential retirements of coal-fired power plants to reduce NO_X emissions in certain nonattainment states would lead to cost increases from the need to use more expensive sources of electricity. This would result in substantial potential reductions in electricity demand. The average potential reduction is 9.8% over the period from 2017 through 2040.

Figure 31: Potential Impacts of a 60 ppb Ozone Standard on Electricity Sector (Sensitivity Case)

	2017	2020	2023	2026	2029	2032	2035	2038
Baseline								
Coal-Fired Capacity (GW)	262	259	257	257	257	257	257	257
Electricity Demand (TWh)	4,150	4,250	4,370	4,480	4,580	4,650	4,750	4,850
Sensitivity Case								
Coal-Fired Capacity (GW)(*)	226	218	153	132	129	125	120	114
Electricity Demand (TWh)	4,060	4,080	3,940	3,900	4,010	4,090	4,170	4,230
Sensitivity Case (Change from Baseline)								
Coal-Fired Capacity (GW)	-37	-41	-104	-125	-128	-132	-137	-143
Electricity Demand (%)	-2.1%	-3.9%	-9.9%	-13%	-12%	-12%	-12%	-13%

Source: NERA calculations as explained in text
(*)In the sensitivity case, due to the high natural gas prices it becomes cost-effective to build new coal-fired generators with carbon capture and sequestration starting in 2032. These new builds are not reflected in the numbers, which are only reflective of the current coal-fired fleet of generators.

3. Potential Impacts on U.S. Sectors

Figure 32 shows the estimated potential changes in sectoral output for the ten sectors in the sensitivity case. The potential reduction in coal output is largely the result of the scrappage of coal-fired power plants in some nonattainment areas. Unlike in the 60 ppb case without natural gas production limitations, the retirement of coal-fired generators does not lead to a large scale increase in natural gas output, because that output is limited due to assumed constraints on new wells development (due to potential permit and offset requirements in nonattainment areas). While agriculture and commercial transportation have the largest potential percentage reductions in non-energy output, the largest absolute reductions over the period from 2017 through 2040 are in the commercial/services and manufacturing sectors. The constraint placed on natural gas production in the sensitivity case would cause additional potential harm to employment, household consumption, and GDP.

Figure 32: Potential Impacts of a 60 ppb Ozone Standard on Sectoral Output (Percentage Changes from Baseline) (Sensitivity Case)

	Avg.	2017	2020	2023	2026	2029	2032	2035	2038
Non-Energy Sectors									
Agriculture	-2.7%	0.4%	-2.2%	-2.4%	-3.0%	-3.5%	-3.7%	-3.6%	-3.3%
Commercial/ Services	-1.2%	-0.4%	-1.1%	-1.1%	-1.3%	-1.4%	-1.5%	-1.4%	-1.4%
Manufacturing	-1.3%	2.0%	-0.6%	-1.3%	-1.7%	-2.0%	-2.3%	-2.4%	-2.2%
Commercial Transportation	-2.4%	-0.9%	-2.4%	-2.2%	-2.6%	-2.8%	-2.8%	-2.7%	-2.5%
Commercial Trucking	-1.5%	-0.1%	-1.5%	-1.5%	-1.7%	-1.9%	-2.0%	-2.0%	-1.9%
Energy Sectors									
Coal	-52%	-10%	-18%	-55%	-66%	-66%	-67%	-67%	-68%
Natural Gas	-11%	0.6%	-1.8%	-7.6%	-11%	-13%	-16%	-18%	-20%
Refining	-2.3%	-0.6%	-1.6%	-2.2%	-2.5%	-2.9%	-3.2%	-2.8%	-2.5%
Crude Oil	0.2%	0.7%	0.3%	0.5%	0.5%	0.1%	-0.1%	-0.2%	-0.1%
Electricity	-9.7%	-2.1%	-3.9%	-10%	-13%	-12.4%	-11.9%	-12%	-13%

Note: Average is the simple average over 2017-2040.
Source: NERA calculations as explained in text

E. Major Uncertainties

We have referred to our results as potential costs and economic impacts as an indication of the major uncertainties involved in developing the estimates. Indeed, as discussed in the next chapter, one of the two major conclusions of the study relates to the need for EPA to develop additional information to reduce these uncertainties as part of its upcoming RIA.

The major uncertainties and data limitations can be summarized in various categories as follows.

- *Baseline future ozone concentrations.* Future ozone concentrations depend upon the nature and location of precursor emissions (as well as on future meteorological conditions). Projections for precursor emissions are inherently uncertain, since they depend on the uncertain future condition of the overall economy as well as on the future circumstances for specific emitting sectors.

- *Identification of nonattainment regions.* The regions in nonattainment of a particular ozone standard depend upon future ozone concentrations as well on the presence of monitoring sites to provide the basis for a non-attainment determination. The number of monitoring sites may increase in the future, with the result that more nonattainment regions may be designated than estimated based on current monitor locations.

- *Emission reductions required to achieve compliance.* The emission reductions needed to achieve compliance depend upon future projected baseline emissions as well as the levels of emissions that are consistent with national compliance. Both of these building blocks are uncertain, leading to uncertainties in the amount and location of necessary emission reductions.

- *Emission controls to obtain the necessary emission reductions.* We have emphasized the limited information on emission controls that is currently available, with EPA's "known" controls representing only one-third of the reductions estimated to be needed to achieve compliance with a 60 ppb standard. We use the available information to develop estimates of the types of sources that could be controlled in order to provide indications of the sources and types of controls that would need to be adopted. But these assessments are highly uncertain.

- *Costs and effectiveness of emission controls.* We have developed an evidence-based approach to determining the likely costs of the "unknown" controls. But these estimates are highly uncertain.

- *Translation of emission control costs into annual costs by sector.* We have allocated our estimates of control costs to different types of costs (capital and O&M) and to different sectors. These allocations also are uncertain, since the mix of capital and O&M may be different than we assume as might be the sectors whose costs would increase.

Although we note that our results are uncertain for these and other reasons, we also point out that we have used the most complete set of available information and have developed an evidence-based approach to fill in for unavailable information. Indeed, we believe that EPA might use the general approach we have developed to update its estimates, although we emphasize that EPA should develop a more complete set of data that would require less extrapolation.

There are of course additional uncertainties involved in modeling the effects of the potential costs related to a 60 ppb ozone standard on the energy sector and the overall economy. The N_{ew}ERA model is a detailed and comprehensive macroeconomic model, but there are of course uncertainties involved in the modeling and the various inputs. The model makes assumptions, for example, about various exogenous factors (e.g., the global price of oil) and about the responsiveness of different productive activities to price changes, all of which could affect the modeling results. Sensitivity analyses are therefore an important element of a full assessment process.

IV. CONCLUSIONS

The conclusions of our study relate to potential impacts of a 60 ppb ozone standard as well as our recommendations for the forthcoming EPA RIA.

A. Potential Impacts of a 60 ppb Ozone Standard

This study has developed estimates of the potential impacts on the U.S. economy, based upon the currently-available information, if the Federal ozone NAAQS were to be set at 60 ppb. The potential impacts would be substantial.

- U.S. GDP potentially would be reduced by about $270 billion per year, or about 1.2% per year over the period from 2017 through 2040.
- The stricter ozone standard would result in a potential average annual loss in consumption per household of about $1,570 over the period 2017 through 2040.
- Labor income reductions would be equivalent to a potential annual reduction of about 2.9 million job-equivalents over the period 2017 through 2040.
- The large changes in NO_X emissions required to achieve compliance would have major potential effects on the U.S. energy sector, including potential retirements of coal-fired units and potential increases in natural gas and electricity prices.
- All sectors of the economy would be negatively affected, with some sectors potentially harmed much more than others.
- All regions of the United States would be negatively affected, with some regions potentially economically disadvantaged much more than others.
- A sensitivity case assuming limits on natural gas production shows that such production constraints could have significant potential implications for U.S. energy markets.
- If, as modeled in the sensitivity case, stricter ozone regulations restricted natural gas production, the potential economic impacts of the stricter ozone standard would be more severe.

These estimated impacts are subject to substantial uncertainties due in part to modeling uncertainties but primarily due to major data limitations. As noted below, we recommend that EPA provide updated emissions, control technology/cost and other information in order to reduce these uncertainties when it develops its RIA.

B. Recommendations for Forthcoming EPA Regulatory Impact Analysis

The large potential costs and macroeconomic impacts found in this study suggest two major recommendations for EPA's forthcoming RIA on its ozone proposal:

1. EPA should develop analyses of the overall costs and economy-wide impacts if it puts forward stricter ozone standards; and

2. EPA should provide updated information on critical parameters, including the potential barriers to crude oil and natural gas production in nonattainment areas as well as updated and expanded estimates of the emission reductions and costs required to achieve alternative ozone standards.

We have developed estimates of the potential impacts of a 60 ppb ozone standard on the U.S. economy and on U.S. households given the available information on emissions and controls, including the impacts of a sensitivity case in which we assume U.S. natural gas production is constrained after 2020 as a result of the ozone standard. It will be important for EPA to provide these types of assessments based upon its estimates of potential compliance costs and resulting impacts on the economy of more stringent ozone standards. It seems clear than a more stringent ozone standard is likely to be very costly and that these compliance costs would have adverse macroeconomic impacts.

It is important that attainment expenditures and macroeconomic impact assessments be based upon reliable information. Our analyses uncovered numerous gaps that EPA should fill as it develops its RIA in order to reduce the large uncertainties in compliance costs and resulting economic impacts. Perhaps the most important gaps are the emission reduction compliance options to achieve the extent of emissions reductions predicted to be needed for attainment, and their costs. The bulk of compliance costs to meet a 60 ppb standard in EPA's 2008-2010 analyses are based upon "unknown controls" (*i.e.*, controls that are not attributed to particular control technologies or even to particular sectors). We develop estimates of these "unknown" costs based upon the best available information and various assumptions, but it would be important for EPA to update its compliance cost information to provide a more comprehensive assessment of emission control options and compliance costs. Moreover, our sensitivity analysis assuming natural gas production constraints shows the importance of this issue for energy markets and the need for EPA to evaluate potential impacts of a tighter ozone standard on domestic energy production, including natural gas and crude oil production.

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APPENDIX A. THE NEWERA MODEL

A. Introduction

NERA developed the NewERA model to forecast the impact of policy, regulatory, and economic factors on the energy sectors and the economy. When evaluating policies that have significant impacts on the entire economy, this model specification captures the effects as they ripple through all sectors of the economy and the associated feedback effects. The NewERA model combines a macroeconomic model with all sectors of the economy with a detailed electric sector model that represents electricity production. This combination allows for a complete understanding of the economic impacts of different policies on all sectors of the economy.

The macroeconomic model incorporates all production sectors except electricity and final demand of the economy. Policy consequences are transmitted throughout the economy as sectors respond until the economy reaches equilibrium. The production and consumption functions employed in the model enable gradual substitution of inputs in response to relative price changes, thus avoiding all-or-nothing solutions.

The main benefit of the integrated framework is that the electric sector can be modeled in great detail yet through integration the model captures the interactions and feedbacks between all sectors of the economy. Electric technologies can be well represented according to engineering specifications. The integrated modeling approach also provides consistent price responses since all sectors of the economy are modeled. In addition, under this framework we are able to model electricity demand response.

The electric sector model is a detailed model of the electric and coal sectors. Each of the more than 17,000 electric generating units in the United States is represented in the model. The model minimizes costs while meeting all specified constraints, such as demand, peak demand, emissions limits, and transmission limits. The model determines investments to undertake and unit dispatch. Because the NewERA model is an integrated model of the entire U.S. economy, electricity demand can respond to changes in prices and supplies. The NewERA model represents the domestic and international crude oil and refined petroleum markets.

The NewERA model outputs include demand and supply of all goods and services, prices of all commodities, and terms of trade effects (including changes in imports and exports). The model outputs also include gross regional product, consumption, investment, and changes in "job equivalents" based on labor wage income, as discussed below in the section on macroeconomic modeling.

B. Overview

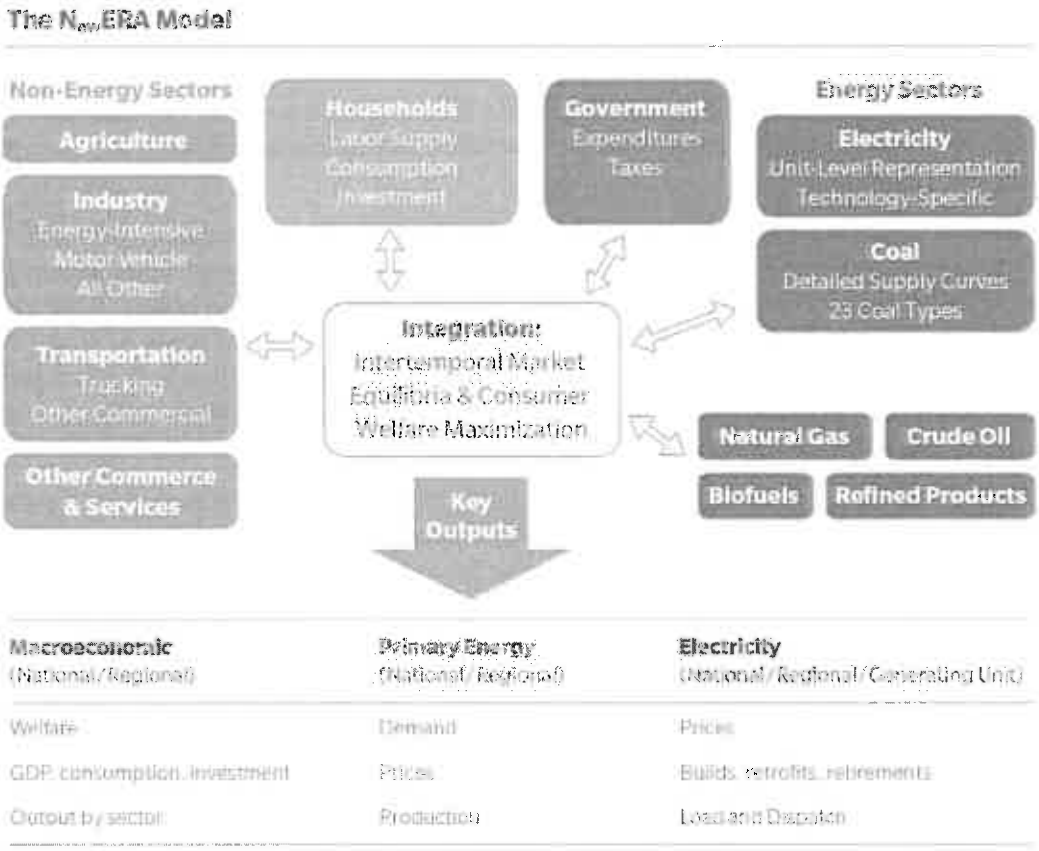
NERA's NewERA modeling system is an integrated energy and economic model that includes a bottom-up representation of the electricity sector, including all of the unit-level details that are

required to accurately evaluate changes in the electric sector. N_{ew}ERA integrates the electricity sector model with a macroeconomic model that includes all other sectors of the economy (except for the electricity production) using a top-down representation. The model produces integrated forecasts for future years; the modeling for this study was for the period from 2014 through 2038 with modeling inputs and results for every third year in this period. The model produces a standard set of reports that includes the following information.

- *Unit-level investments in the electric sector* – retrofits in response to environmental policies, new builds (full range of new generation technologies represented), retirements based on economics.
- *Prices* – wholesale electricity prices for each of 34 U.S. regions, capacity prices for each U.S. region, delivered electricity prices by sector for each of 11 macroeconomic regions in N_{ew}ERA, Henry Hub natural gas prices and delivered natural gas prices to the electric sector for each U.S. region, minemouth coal prices for 24 different types of coal, delivered coal prices by coal unit, refined oil product prices (gasoline and diesel fuel), renewable energy credit (REC) prices for each state/regional renewable portfolio standard (RPS), and emissions prices for all regional and national programs with tradable credits.
- *Macroeconomic results* – gross domestic product (and gross regional product for each macroeconomic region), welfare, changes in disposable income, and changes in labor income and real wage rates (used to estimate labor market changes in terms of an equivalent number of jobs).

Figure A-1 provides a simplified representation of the key elements of the N_{ew}ERA modeling system.

Figure A-1: N_{ew}ERA Modeling System Representation



C. Electric Sector Model

The electric sector model that is part of the N_{ew}ERA modeling system is a bottom-up model of the electric and coal sectors. Consistent with the macroeconomic model, the electric sector model is fully dynamic and includes perfect foresight (under the assumption that future conditions are known). Thus, all decisions within the model are based on minimizing the present value of costs over the entire time horizon of the model while meeting all specified constraints, including demand, peak demand, emissions limits, transmission limits, RPS regulations, fuel availability and costs, and new build limits. The model set-up is intended to mimic (as much as is possible within a model) the approach that electric sector investors use to make decisions. In determining the least-cost method of satisfying all these constraints, the model endogenously decides:

- What investments to undertake (*e.g.*, addition of retrofits, build new capacity, repower unit, add fuel switching capacity, or retire units);

- How to operate each modeled unit (*e.g.*, when and how much to operate units, which fuels to burn) and what is the optimal generation mix; and

- How demand will respond. The model thus assesses the trade-offs between the amount of demand-side management (DSM) to undertake and the level of electricity usage.

Each unit in the model has certain actions that it can undertake. For example, all units can retire, and many can undergo retrofits. Any publicly-announced actions, such as planned retirements, planned retrofits (for existing units), or new units under construction can be specified. Coal units have more potential actions than other types of units. These include retrofits to reduce emissions of SO_2, NO_X, mercury, and CO_2.[30] The costs, timing, and necessity of retrofits may be specified as scenario inputs or left for the model to endogenously select. Coal units can also switch the type of coal that they burn (with practical unit-specific limitations). Finally, coal units may retire if none of the above actions will allow them to remain profitable, after accounting for their revenues from generation and capacity services.

Most of the coal units' actions would be in response to environmental limits that can be added to the model. These include emission caps (for SO_2, NO_X, Hg, and CO_2) that can be applied at the national, regional, state or unit level. We can also specify allowance prices for emissions, emission rates (especially for toxics such as Hg) or heat rate levels that must be met. For this analysis, we have assumed that retirements of existing coal-fired generators in some states are part of the compliance actions of those states to achieve targeted NO_X reductions.

Just as with investment decisions, the operation of each unit in a given year depends on the policies in place (*e.g.*, unit-level standards), electricity demand, and operating costs, especially energy prices. The model accounts for all these conditions in deciding when and how much to operate each unit. The model also considers system-wide operational issues such as environmental regulations, limits on the share of generation from intermittent resources, transmission limits, and operational reserve margin requirements in addition to annual reserve margin constraints.

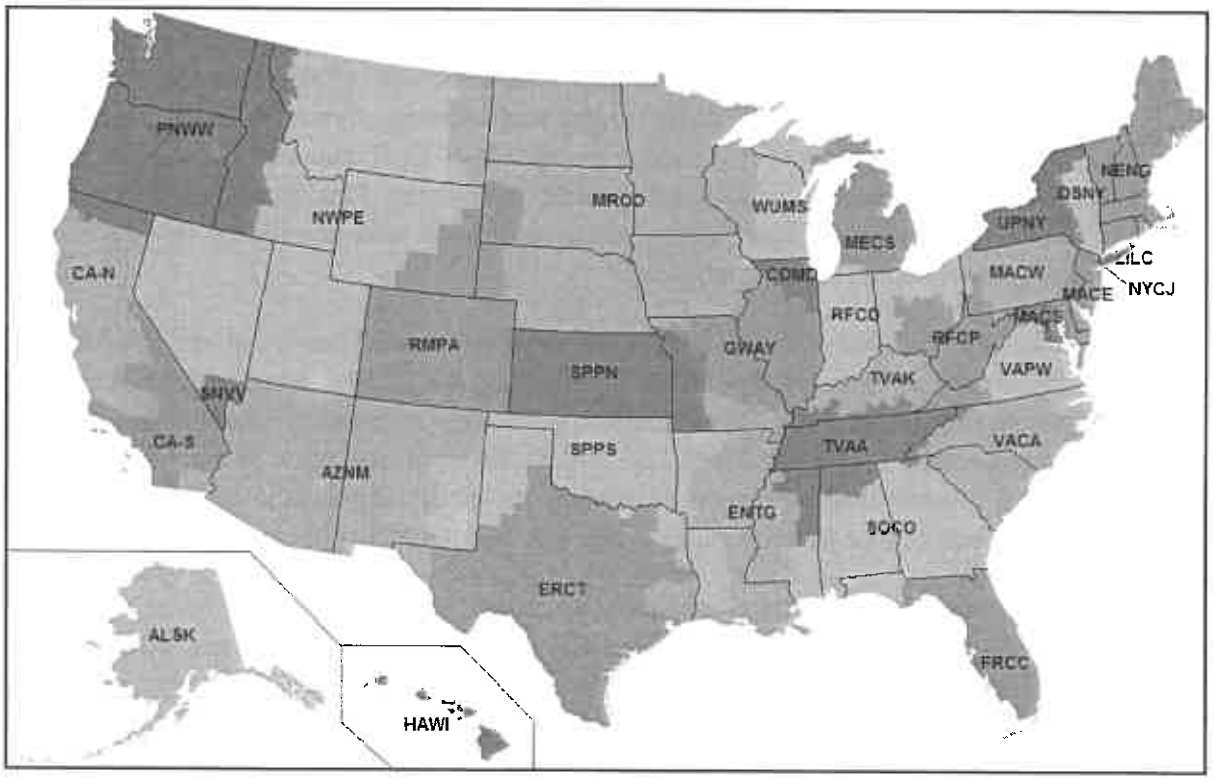
To meet increasing electricity demand and reserve margin requirements over time, the electric sector must build new generating capacity. Future environmental regulations and forecasted energy prices influence which technologies to build and where. For example, if a national RPS policy is to take effect, some share of new generating capacity will need to come from renewable power. On the other hand, if there is a policy to address emissions, it might elicit a response to retrofit existing fossil-fired units with pollution control technology or enhance existing coal-fired units to burn different types of coals, biomass, or natural gas. Policies calling for improved heat rates may lead to capital expenditure spent on repowering existing units. All of these policies

[30] As discussed in the report body, N_{ew}ERA does not incorporate EPA's recently proposed power sector CO_2 rule.

will also likely affect retirement decisions. The N_{ew}ERA electric sector model endogenously captures all of these different types of decisions.

The model contains 34 U.S. electricity regions (and six Canadian electricity regions). Figure A-2 shows the U.S. electricity regions.

Figure A-2: N_{ew}ERA Electric Sector Model – U.S. Regions



The electric sector model is fully flexible in the model horizon and the years for which it solves. When used in an integrated manner with the macroeconomic model, and to analyze long-term effects, the model has the same time steps as in the macroeconomic model (2014 through 2038, modeling every third year).

D. Macroeconomic Model

1. Overview

The N_{ew}ERA macroeconomic model is a forward-looking dynamic computable general equilibrium (CGE) model of the United States. The model simulates all economic interactions in the U.S. economy, including those among industry, households, and the government. Additional background information on CGE models can be found in Burfisher (2011).

The N_{ew}ERA CGE framework uses the standard theoretical macroeconomic structure to capture the flow of goods and factors of production within the economy. A simplified version of these interdependent macroeconomic flows is shown in Figure A-3. The model implicitly assumes "general equilibrium," which implies that all sectors in the economy are in balance and all economic flows are endogenously accounted for within the model. In this model, households supply factors of production, including labor and capital, to firms. Firms provide households with payments for the factors of production in return. Firm output is produced from a combination of productive factors and intermediate inputs of goods and services supplied by other firms. Individual firm final output can be consumed within the United States or exported. The model also accounts for imports into the United States. In addition to consuming goods and services, households can accumulate savings, which they provide to firms for investments in new capital. Government receives taxes from both households and firms, contributes to the production of goods and services, and also purchases goods and services. Although the model assumes equilibrium, a region in the model can run deficits or surpluses in current accounts and capital accounts. In aggregate, all markets clear, meaning that the sum of regional commodities and factors of production must equal their demands, and the income of each household must equal its factor endowments plus any net transfers received.

The model uses the standard CGE framework developed by Arrow and Debreu (1954). Behavior of households is represented by a nested Constant Elasticity of Substitution (CES) utility function. The model assumes that households seek to maximize their overall welfare, or utility, across time periods. Households have utility functions that reflect trade-offs between leisure (which reduces the amount of time available for earning income) and an aggregate consumption of goods and services. Households maximize their utility over all time periods subject to an intertemporal budget constraint based on their income from supplying labor, capital, and natural resource to firms. In each time period, household income is used to consume goods and services or to fund investment. Within consumption, households substitute between energy (including electricity, coal, natural gas, and petroleum), personal transportation, and goods and services based on the relative price of these inputs. Figure A-4 illustrates the utility function of the households.

Figure A-3: Interdependent Economic Flows in N_{ew}ERA's Macroeconomic Model

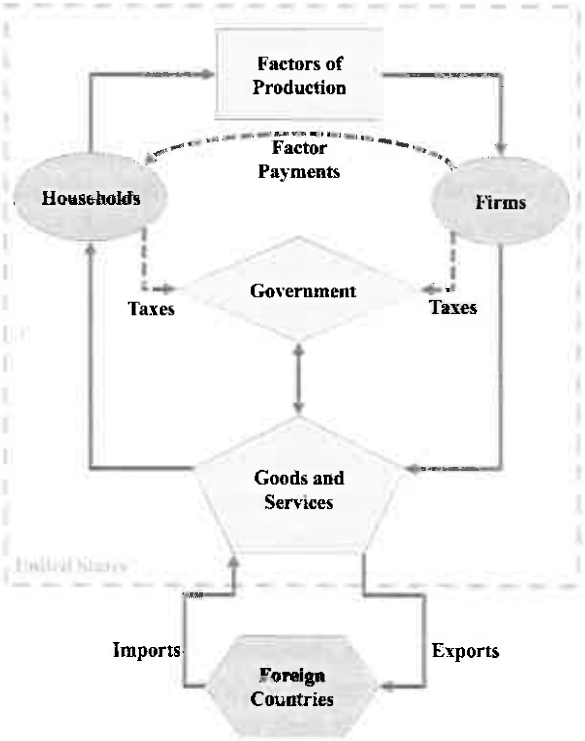
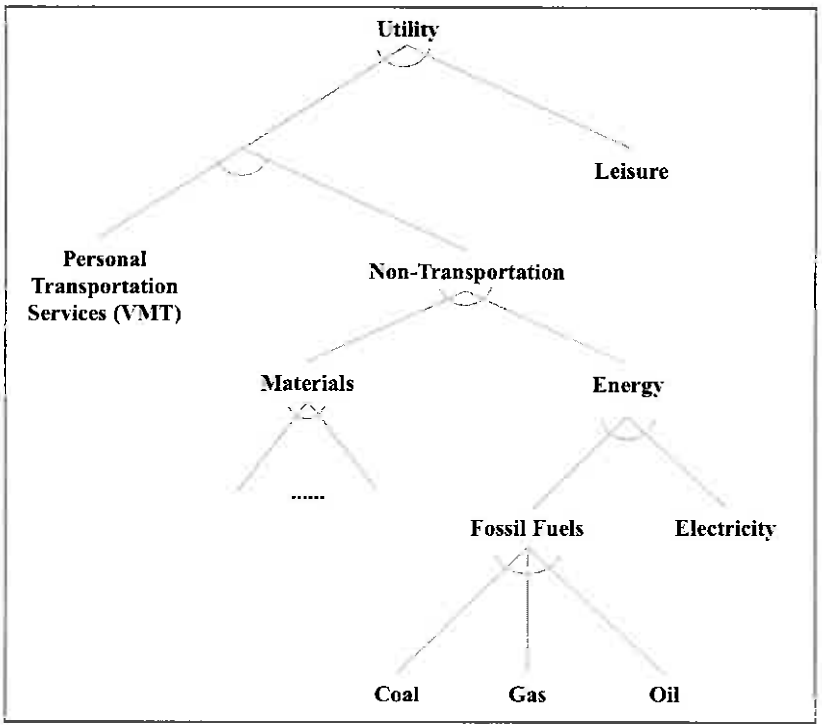


Figure A-4: Household Consumption Structure in N_{ew}ERA's Macroeconomic Model



On the production side, Figure A-5 shows the production structure of the commercial transportation and the trucking sector. Production structure for the rest of the industries is shown in Figure A-6. The model assumes all industries maximize profits subject to technological constraints. The inputs to production are energy (including the same four types noted above for household consumption), capital, and labor. Production also uses inputs from intermediate products (*i.e.*, materials) provided by other firms. The N_{ew}ERA model allows producers to change the technology and the energy source they use to manufacture goods. If, for example, petroleum prices rise, an industry can shift to a cheaper energy source. It can also choose to use more capital or labor in place of petroleum, increasing energy efficiency and maximizing profits with respect to industry constraints.

Figure A-5: Commercial Transportation and Trucking Sector Production Structure in N_{ew}ERA's Macroeconomic Model

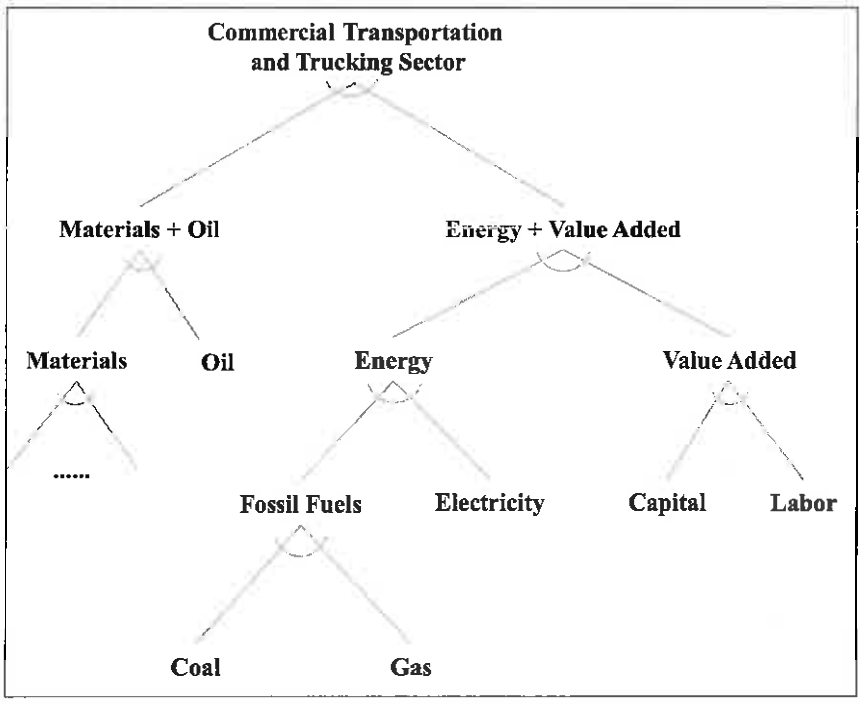
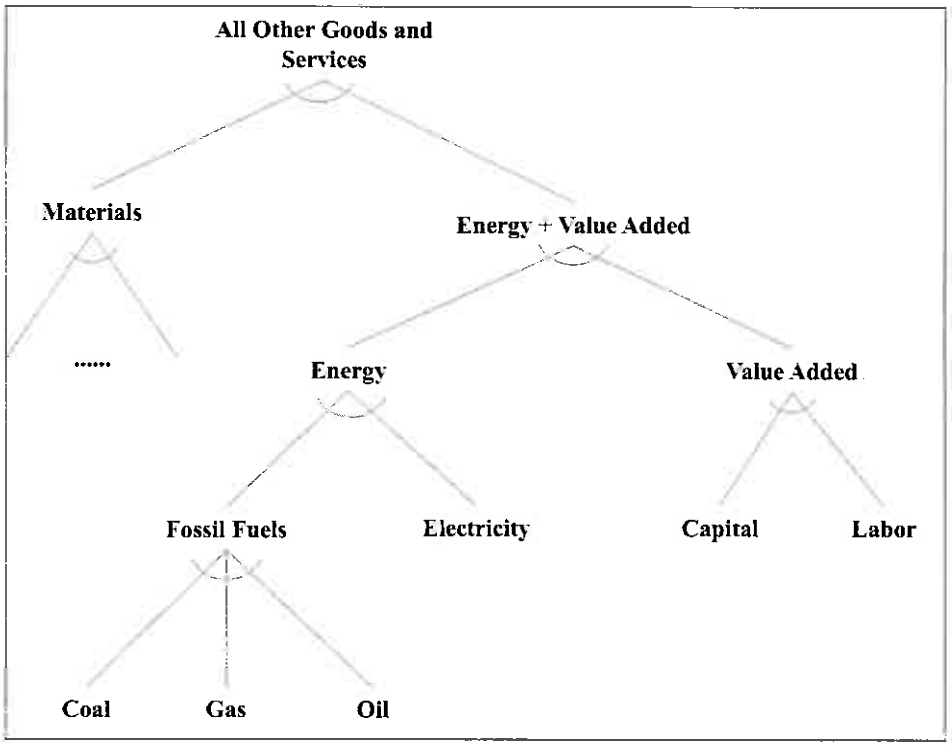


Figure A-6: Production Structure for Other Sectors in N_{ew}ERA's Macroeconomic Model



All goods and services, except crude oil, are treated as Armington goods, which assume the domestic and foreign goods are differentiated and thus are imperfect substitutes (Armington 1969). The level of imports depends upon the elasticity of substitution between the imported and domestic goods. The Armington elasticity among imported goods is assumed to be twice as large as the elasticity between the domestic and imported goods, characterizing the greater substitutability among imported goods.

Business investment decisions are informed by future policies and outlook. The forward-looking characteristic of the model enables businesses and consumers to determine the optimal savings and investment levels while anticipating future policies with perfect foresight.

The benchmark year economic interactions are based on the IMPLAN 2008 database, which includes regional detail on economic interactions among 440 different economic sectors. The macroeconomic and energy forecasts that are used to project the benchmark year going forward are calibrated to EIA's *Annual Energy Outlook* (*AEO*) *2014* Reference case.

2. Interactions between Compliance Costs, Capital Investment, and Household Expenditures

Regulations cause producers in the affected industries to make capital expenditures that they would not make otherwise. In addition, regulations change consumption patterns for households. To model the macroeconomic impacts of regulations, N_{ew}ERA accounts for interactions between compliance costs, capital investments, and household expenditures based on the following three effects.

1. *Compliance costs for producers in the regulated industries.* Producers in the regulated industries have to make capital expenditures to comply with the regulation. These expenditures increase the costs of producing goods and services in the regulated industries. The higher costs lead to higher prices for the goods and services, which in turn lead to lower demand in the regulated industries. Thus, this effect reduces economic activity.

2. *Scarcity effect due to non-optimal capital allocation.* In N_{ew}ERA's modeling framework, the capital expenditures for regulatory compliance are assumed to be unproductive. The capital expenditures in the regulated industries make less capital available to produce goods and services throughout the economy. In other words, the unproductive capital expenditures in the regulated industries "crowd out" productive capital investment in the broader economy. This scarcity effect increases the opportunity cost of capital in the economy, which implies higher costs of capital. This in turn lowers investment in productive capital and slows economic growth.

3. *Household purchases of unproductive durable goods.* Regulations also cause households to change their consumption patterns, particularly in terms of durable goods. For example, households may need to purchase new automobiles, lawn mowers, or equipment for compliance with the regulation. These additional expenditures on unproductive durable goods are non-optimal from the standpoint of households, but they represent increased demand for the manufacturing sector. Thus, these additional household purchases increase economic activity.

The net macroeconomic impacts of regulations calculated by N_{ew}ERA reflect the combination of these three effects.

3. Regional Aggregation

The N_{ew}ERA macroeconomic model includes 11 regions built up from economic data for the 50 U.S. states and the District of Columbia. The regions are shown in Figure A-7.

Figure A-7: N_{ew}ERA Macroeconomic Model Regions

4. Sectoral Aggregation

The N_{ew}ERA model includes a standard set of 10 economic sectors: five energy (coal, natural gas, crude oil, electricity, and refined petroleum products) and five non-energy sectors (services, manufacturing, agriculture, commercial transportation excluding trucking, and trucking). These sectors are aggregated up from the 440 IMPLAN sectors. The model has the flexibility to represent sectors at different levels of aggregation, when warranted, to better meet the needs of specific analyses.

5. Natural Gas and Crude Oil Markets

As with most commodity markets, there are uncertainties about how the U.S. natural gas market will evolve, and the N_{ew}ERA modeling system is designed explicitly to address the key factors affecting future natural gas supply and prices. To account for natural gas supply uncertainty and the subsequent effect it could have on international markets, the N_{ew}ERA modeling system has the ability to represent supply curves for conventional natural gas and shale gas for each region

of the model. By including each type of natural gas, it is possible to incorporate expert judgments and sensitivity analyses on a variety of uncertainties, such as the extent of shale gas reserves, the cost of shale gas production, and the impacts of environmental regulations.

The N_{ew}ERA model represents the domestic and international crude oil and refined petroleum markets. The international markets are represented by flat supply curves with exogenously specified prices. Because crude oil is treated as a homogeneous good, the international price for crude oil sets the U.S. price for crude oil.

For this study, we calibrated natural gas and crude oil production at the state level based on information from *AEO 2014*. While *AEO 2014* does not provide state-level information, they did provide us with basin-specific production forecasts that we translated into state-level production based on historical state-level production, other publicly-available forecasts by state, and our own expertise.

6. Macroeconomic Outputs

As with other CGE models, the N_{ew}ERA macroeconomic model outputs include demand and supply of all goods and services, prices of all commodities, and terms of trade effects (including changes in imports and exports). The model outputs also include gross regional product, consumption, investment, cost of living or burden on consumers, and changes in "job equivalents" based on changes in labor wage income. All model outputs are calculated by time, sector, and region.

Impacts on workers are often considered an important output of policy evaluations. Impacts on workers are complicated to estimate and to explain because they can include several different impacts, including involuntary unemployment, reductions in wage rates for those who continue to work, and voluntary reductions in hours worked due to lower wage rates. No model addresses all of these potential impacts. The N_{ew}ERA model is a long-run equilibrium model based upon full employment, and thus its results relate to the longer-term effects on labor income and voluntary reductions in hours worked rather than involuntary unemployment impacts. It addresses long-run employment impacts, all of which are based on estimates of changes in labor income, also called the "wage bill" or "payments to labor." Labor income impacts consist of two effects: (1) changes in real wage per hour worked; and (2) changes in labor market participation (hours worked) in response to changed real wage rates. The labor income change can also be expressed on a per-household basis, which represents one of the key components of disposal income per household. (The other key components of disposable income are returns on investments or "payments to capital," and income from ownership of natural resources). The labor income change can also be stated in terms of job-equivalents, by dividing the labor income change by the annual income from the average job. A loss of one job-equivalent does not necessarily mean one less employed person—it may be manifested as a combination of fewer people working and less income per person who is working. However, this measure allows us to

express employment-related impacts in terms of an equivalent number of employees earning the average prevailing wage.

For modeling the economic impacts of changes in energy prices, we assume that 50% of the wealth impacts would accrue to local residents in each energy production region (state), and the remaining 50% of wealth impacts would accrue to energy company shareholders based on national population percentages. We are not aware of any recent studies of the geographic distribution of potential energy sector gains, so we used an even division between state and national impacts given that some energy companies are in-state and some gains to national companies would accrue to local residents. A large fraction of energy production (particularly for natural gas shale developments that have become available through horizontal drilling techniques and hydraulic fracturing, or "fracking") is on private land and generates payments to local residents (payments, severance taxes, renegotiated leases, etc.). The remaining wealth impacts from changes in energy prices would affect shareholders in large publicly-traded energy companies, who are spread throughout the country.

E. Integrated N_{ew}ERA Model

The N_{ew}ERA modeling framework fully integrates the macroeconomic model and the electric sector model so that the final solution is a consistent equilibrium for both models and thus for the entire U.S. economy.

To analyze any policy scenario, the system first solves for a consistent baseline solution; it then iterates between the two models to find the equilibrium solution for the scenario of interest. For the baseline, the electric sector model is solved first under initial economic assumptions and forecasts for electricity demand and energy prices. The equilibrium solution provides the baseline electricity prices, demand, and supply by region as well as the consumption of inputs—capital, labor, energy, and materials—by the electric sector. These solution values are passed to the macroeconomic model.

Using these outputs from the electric sector model, the macroeconomic model solves the baseline while constraining the electric sector to replicate the solution from the electric sector model and imposing the same energy price forecasts as those used to solve the electric sector baseline. In addition to the energy price forecasts, the macroeconomic model's non-electric energy sectors are calibrated to the desired exogenous forecast (EIA's *AEO 2014* forecast) for energy consumption, energy production, and macroeconomic growth. The macroeconomic model solves for equilibrium prices and quantities in all markets subject to meeting these exogenous forecasts.

After solving the baseline, the integrated N_{ew}ERA modeling system solves for the scenario. First the electric sector model reads in the scenario definition. The electric sector model then solves for the equilibrium level of electricity demand, electricity supply, and inputs used by the electric sector (*i.e.*, capital, labor, energy, emission permits). The electric sector model passes these

equilibrium solution quantities to the macroeconomic model, which solves for the equilibrium prices and quantities in all markets. The macroeconomic model then passes to the electric sector model the following (solved for equilibrium prices):

- Electricity prices by region;
- Prices of non-coal fuels used by the electric sector (*e.g.*, natural gas and oil); and
- Prices of any permits that are tradable between the non-electric and electric sectors (*e.g.*, carbon permits under a nationwide greenhouse gas cap-and-trade program).

The electric sector model then solves for the new electric sector equilibrium, taking the prices from the macroeconomic model as exogenous inputs. The models iterate—prices being sent from the macroeconomic model to the electric sector model and quantities being sent from the electric sector model to the macroeconomic model—until the prices and quantities in the two models differ by less than a fraction of a percent.

This decomposition algorithm allows the N_{ew}ERA model to retain the information in the detailed electricity model, while at the same time accounting for interactions with the rest of the economy. The detailed information on the electricity sector enables the model to represent regulatory policies that are imposed on the electricity sector in terms of their impacts at a unit level.

F. References to the Appendix

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APPENDIX B. ESTIMATES OF STATE-SPECIFIC NO_X EMISSIONS REDUCTIONS REQUIRED TO ACHIEVE 60 PPB OZONE STANDARD

This appendix provides state-specific estimates of the NO_X reductions required to achieve compliance with a 60 ppb ozone standard. The estimates are based upon EPA projected 2018 baseline emissions and estimates of the emission levels that would achieve the 60 ppb standard. The baseline NO_X emissions are based on EPA's most recent projections for the furthest year in the future (2018), supplemented by N_{ew}ERA results for EGU emissions. The estimated NO_X emissions consistent with a 60 ppb ozone standard are based upon EPA's 2008-2010 ozone analyses.[31]

A. Baseline NO_X Emission Projections

1. State-Level Baseline Emissions Projections

The following table shows the baseline NO_X emission projections by state used in our analysis.[32] As shown in the table, baseline NO_X emission projections for point (non-EGU), area (non-point), onroad mobile, and nonroad mobile sources reflect EPA's most recent projections for 2018, which are based on historical emissions data for 2011 (EPA 2014a). Baseline NO_X emission projections for EGUs reflect N_{ew}ERA baseline outputs for 2026. As discussed in Appendix A, N_{ew}ERA incorporates a detailed database of all power plants in the United States, including their emission rates and operational characteristics. The EPA information for 2018 includes EGU emission projections, and its projections are generally similar to those from N_{ew}ERA; we use N_{ew}ERA EGU emission projections for our analysis to maintain internal consistency with our economic impact modeling using N_{ew}ERA. The EGU baseline NO_X emission projections from N_{ew}ERA reflect outputs for 2026 because compliance deadlines for a new ozone standard of 60 ppb would presumably be around that year for most states, as discussed in Appendix D. Note that we use EPA's projections for the furthest year in the future (2018) as estimates for emissions in the compliance year; we expect EPA to develop updated emissions information when it releases its ozone proposal.

[31] "EPA's 2008-2010 ozone analyses" refers to information in EPA's 2008 regulatory impact analysis for the ozone NAAQS, including information on baseline future conditions and ozone standards of 84 and 75 ppb (EPA 2008); EPA's 2010 supplemental regulatory impact analysis, including information on an ozone standard of 60 ppb (EPA 2010); and data files in Docket No. EPA-HQ-OAR-2007-0225.

[32] Our analysis does not include Alaska or Hawaii, because EPA did not model air quality in these states in its 2008-2010 ozone analyses. Our analysis also excludes the District of Columbia.

Table B-1. Baseline NO_X Emission Projections by State and Source Category (1000s of tons)

	EPA				N_{ew}ERA	NERA Baseline
	Point	Area	Onroad	Nonroad	EGU	Total
U.S. Total	**1,784**	**1,680**	**2,645**	**2,071**	**1,524**	**9,705**
Alabama	60	25	61	33	83	262
Arizona	16	7	53	37	38	152
Arkansas	33	17	38	30	20	138
California	74	66	246	160	8	553
Colorado	52	45	41	30	36	203
Connecticut	5	13	15	15	1	49
Delaware	2	2	6	8	<1	20
Florida	52	25	134	106	41	358
Georgia	51	20	110	47	17	245
Idaho	11	7	25	16	<1	58
Illinois	69	58	73	94	53	347
Indiana	65	27	67	47	114	319
Iowa	30	18	29	54	42	174
Kansas	50	69	27	54	16	217
Kentucky	32	46	49	33	84	245
Louisiana	120	97	46	122	21	407
Maine	13	11	10	9	<1	44
Maryland	16	13	42	24	15	110
Massachusetts	14	22	28	27	6	97
Michigan	58	65	84	48	65	321
Minnesota	32	34	53	53	23	195
Mississippi	39	11	33	23	20	126
Missouri	31	18	125	56	40	269
Montana	7	19	12	30	22	90
Nebraska	13	9	19	79	34	153
Nevada	9	4	26	15	9	63
New Hampshire	2	5	9	5	2	22
New Jersey	12	24	31	34	5	106
New Mexico	24	51	31	25	14	146
New York	41	75	92	69	14	291
North Carolina	38	25	101	40	39	243
North Dakota	10	17	9	34	57	127
Ohio	58	40	134	64	96	392
Oklahoma	79	97	44	32	60	312
Oregon	15	14	33	28	1	92
Pennsylvania	62	105	100	52	52	371
Rhode Island	1	6	4	3	<1	15
South Carolina	26	12	46	25	14	123
South Dakota	3	7	9	19	1	39
Tennessee	39	30	65	35	29	199
Texas	214	263	213	149	121	959
Utah	20	31	33	13	57	154
Vermont	<1	4	5	3	0	12
Virginia	38	31	68	47	22	207
Washington	24	10	85	61	7	187
West Virginia	25	53	17	15	66	176
Wisconsin	31	23	52	37	18	162
Wyoming	65	4	12	33	41	155

Note: EPA emission projections are for 2018, and N_{ew}ERA EGU emission projections represent 2026.
Source: NERA calculations as explained in text

2. Oil and Gas Production Projections

EPA's recent 2018 NO_X emission projections for oil and gas activity are based on oil and gas production projections from the Energy Information Administration's *Annual Energy Outlook (AEO) 2013* (EPA 2014b, p. 104). As discussed in Appendix A, we calibrated future oil and gas production in N_{ew}ERA to *AEO 2014*. *AEO 2014* has a 22% higher national crude oil production projection and an 11% higher national natural gas production projection for 2019 than *AEO 2013* (EPA 2014b, p. 104). Our use of EPA emission projections for oil and gas activity based on the *AEO 2013* thus somewhat understates baseline future emissions and thus necessary emission reductions for 60 ppb.

B. NO_X Compliance Emissions

1. State-Level Compliance Emissions

We used state-specific information from EPA's 2008-2010 ozone analyses of baseline (existing standard) NO_X emission projections (baseline emissions) and NO_X reduction requirements to estimate NO_X emission levels in each state consistent with compliance with 60 ppb (compliance emissions). The necessary NO_X emission reductions for 60 ppb in each state are the difference between baseline emissions and compliance emissions.

The EPA baseline emissions projections represent 2020 and are based on historical data from 2002. Through a series of modeling exercises, EPA estimated NO_X emission reductions by sub-state area that would be needed for compliance with several alternative national ozone standards, including 60 ppb. We gathered EPA information on these reductions and aggregated baseline emissions and reduction requirements to the state level. EPA methodology was based upon noncompliance areas using its then-existing ozone monitoring information. EPA did assume that counties adjacent to nonattainment counties would need to reduce emissions to achieve compliance. But EPA did not consider the possibility of noncompliance for counties without monitors; this omission tends to reduce the extent of potential noncompliance, which would also reduce the potential costs and economic impacts of compliance with a national 60 ppb standard.

The EPA information from the 2008-2010 ozone analyses indicated that some areas of California and Texas would not comply with the 1997 ozone standard of 0.80 ppm (84 ppb based on averaging convention) under baseline conditions. The information also indicated that areas of several states would require reductions to achieve 75 ppb, which became the new ozone standard in the 2008-2010 NAAQS review.

To estimate compliance emissions in each state for past, current, and potential future ozone standards (84 ppb, 75 ppb, and 60 ppb), we subtracted any reductions required to meet each standard in EPA's analysis from 2020 baseline emissions. This calculation and the resulting compliance emissions by state are shown in Table B-2. When a state did not require any emission reductions in EPA's analysis to meet a certain standard (*e.g.*, Georgia to comply with a 75 ppb standard), we were not able to estimate state compliance emissions for that standard; in such cases, compliance emissions could be greater than or equal to 2020 baseline emissions.

Table B-2. EPA Information from 2008-2010 Ozone Analyses: Baseline NO_X Emissions and Necessary NO_X Emission Reductions for 84, 75, and 60 ppb (1000s tons of NO_X)

	2020 Baseline	Reductions from 2020 Baseline to 84ppb	84ppb Compliance Emissions	Reductions from 84ppb to 75ppb	75ppb Compliance Emissions	Reductions from 75ppb to 60ppb	60ppb Compliance Emissions
U.S. Total	**10,728**	**816**	**N/A**	**582**	**N/A**	**3,367**	**N/A**
Alabama	226	-	-	-	-	38	188
Arizona	159	-	-	-	-	50	109
Arkansas	145	-	-	-	-	34	111
California	727	576	151	45	106	49	56
Colorado	185	-	-	-	-	63	122
Connecticut	51	-	-	14	37	20	18
Delaware	36	-	-	6	30	17	13
Florida	392	-	-	-	-	63	328
Georgia	281	-	-	-	-	107	174
Idaho	70	-	-	-	-	17	53
Illinois	412	-	-	101	310	157	154
Indiana	316	-	-	69	247	122	126
Iowa	203	-	-	-	-	7	197
Kansas	240	-	-	-	-	34	205
Kentucky	219	-	-	-	-	80	139
Louisiana	517	-	-	-	-	367	151
Maine	47	-	-	-	-	16	31
Maryland	119	-	-	31	89	48	40
Massachusetts	115	-	-	-	-	50	66
Michigan	378	-	-	-	-	221	158
Minnesota	294	-	-	-	-	-	-
Mississippi	185	-	-	-	-	58	127
Missouri	283	-	-	-	-	106	178
Montana	87	-	-	-	-	-	-
Nebraska	151	-	-	-	-	4	146
Nevada	80	-	-	-	-	19	60
New Hampshire	34	-	-	-	-	2	31
New Jersey	138	-	-	31	108	57	50
New Mexico	178	-	-	-	-	69	110
New York	303	-	-	53	250	136	114
North Carolina	218	-	-	-	-	111	107
North Dakota	100	-	-	-	-	-	-
Ohio	391	-	-	-	-	238	152
Oklahoma	300	-	-	-	-	55	245
Oregon	129	-	-	-	-	12	117
Pennsylvania	348	-	-	72	276	151	125
Rhode Island	13	-	-	-	-	2	11
South Carolina	150	-	-	-	-	83	66
South Dakota	36	-	-	-	-	1	36
Tennessee	235	-	-	-	-	73	161
Texas	1,147	239	907	109	798	311	488
Utah	126	-	-	-	-	34	91
Vermont	12	-	-	-	-	-	-
Virginia	250	-	-	-	-	110	140
Washington	209	-	-	-	-	49	160
West Virginia	147	-	-	-	-	54	92
Wisconsin	213	-	-	51	161	53	109
Wyoming	133	-	-	-	-	17	116

Note: "-" denotes that all areas of the state would comply with the ozone standard under 2020 baseline conditions according to EPA information in the 2008-2010 ozone analyses.
"N/A" denotes that U.S. total NO_X emissions for compliance with 84, 75, or 60 ppb are not applicable because not all states have estimated compliance emission levels for each ozone standard.

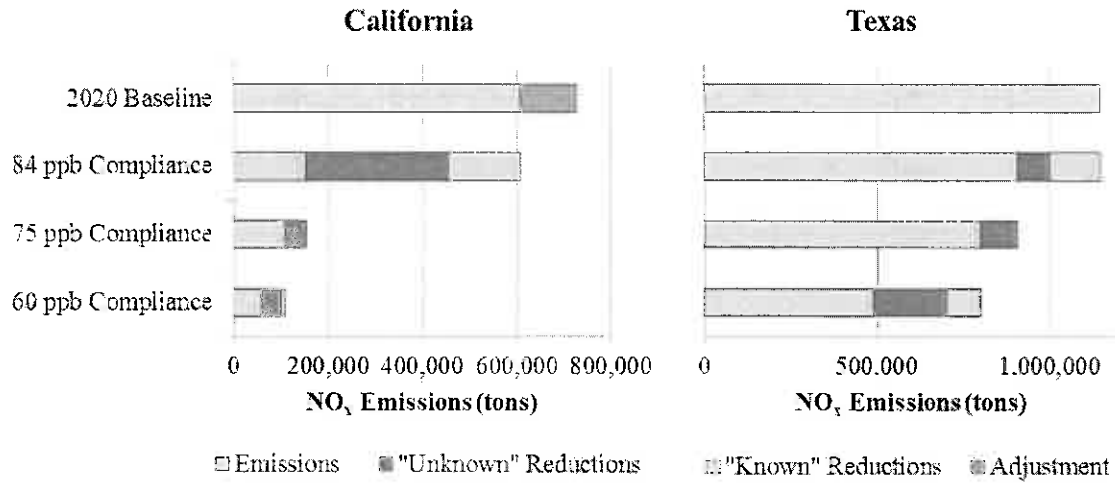
Source: EPA 2008-2010 ozone analyses and NERA calculations as explained in text

2. Texas and California Compliance Emissions

According to EPA's 2008-2010 ozone analyses, the Los Angeles-South Coast and San Joaquin Valley areas in California and the Houston area in Texas would have baseline ozone levels above the 1997 standard of 84 ppb in the future year analyzed by EPA (2020). EPA's 2008-2010 ozone analysis data includes information on baseline future emissions and necessary future emission reductions in these areas for compliance with 84 ppb, as well as necessary future emission reductions to comply with new ozone standards, including 75 and 60 ppb. In the 2008-2010 ozone analyses, EPA assumed that the California areas, which are the only ozone non-attainment areas in the country classified as Extreme, would have a longer timeline for compliance with new ozone standards, and EPA excluded the compliance costs for the California areas from the main compliance cost analysis in its 2008-2010 analyses.

The following figure summarizes our understanding of EPA data for California and Texas from the 2008-2010 analyses on baseline future emissions and necessary emission reductions for compliance with 84, 75, and 60 ppb. The figure presents our attempt at assembling information for these two states, information that is not as clear in the EPA docket files as the information for other states (with baseline future ozone levels below 84 ppb).

Figure B-1. Estimating Compliance Emissions in California and Texas



Note: EPA modeled the Los Angeles-South Coast and San Joaquin Valley regions in 2030 (as opposed to 2020, the analysis year for other states). EPA included 120,000 tons of NO_X emission reductions to account for inventory changes between 2020 and 2030 attributable to recent locomotive-marine regulations (EPA 2008, p. 7b-2): we treat these reductions as a baseline adjustment.
Nearly all of the "known" reductions specifically for 60 ppb compliance occur outside of the extrapolation areas with severe non-attainment problems (*i.e.*, in areas projected to comply with 75 ppb but not with 60 ppb).

Source: NERA calculations as explained in text

C. Required NO_X Emission Reductions

Our analysis assumes that the compliance emissions for different standards implied by EPA's 2008-2010 ozone analyses would remain constant throughout the future. Using our updated baseline emissions described above, we estimated necessary NO_X emission reductions for different ozone standards as the difference between updated baseline emission projections and compliance emissions for the different standards. These required emission reduction estimates for standards of 80 ppm (84 ppb based on averaging convention), 75 ppb, and 60 ppb in each state are summarized in Table B-3.

In several cases, EPA projected in its 2008-2010 ozone analyses that a state would be compliant with a 75 ppb or 60 ppb standard, but the state's updated baseline emission projection for 2018 used in our analysis (based on 2011 data) is higher than the 2020 baseline emission projection used in EPA's 2008-2010 analysis (based on 2002 data). We were unable to infer compliance emissions levels for these states and standards from EPA's analysis (since EPA did not require any reductions from baseline emissions), so we reviewed recent historical NO_X emissions and ozone concentrations to judge whether these states were still likely to be in compliance even with the higher baseline emissions in the updated projection used in our analysis. In each of these cases except the current standard of 75 ppb in Colorado, we judged that states estimated to be in attainment in EPA's 2008-2010 analyses would still be in attainment even with higher baseline NO_X emissions. Colorado had at least one ozone monitor exceeding 75 ppb in all recent years, so we assumed it would need to return to its projected baseline emissions level in EPA's 2008-2010 ozone analyses in order to be in attainment for 75 ppb.

Table B-3. 2018 Baseline Emission Projections and Reduction Requirements for Ozone Standards (1000s tons of NO_X)

	Updated Baseline Emission Projections	Compliance Emissions			Reductions Required (from Updated Baseline Emissions)		
		84ppb	75ppb	60ppb	84ppb	75ppb	60ppb
U.S. Total	**9,705**	**N/A**	**N/A**	**N/A**	**454**	**904**	**3,866**
Alabama	262	-	-	188	-	-	75
Arizona	152	-	-	109	-	-	43
Arkansas	138	-	-	111	-	-	27
California	553	151	106	56	403	448	497
Colorado	203	-	-	122	-	18	81
Connecticut	49	-	37	18	-	11	31
Delaware	20	-	30	13	-	-	7
Florida	358	-	-	328	-	-	30
Georgia	245	-	-	174	-	-	70
Idaho	58	-	-	53	-	-	5
Illinois	347	-	310	154	-	37	194
Indiana	319	-	247	126	-	72	193
Iowa	174	-	-	197	-	-	-
Kansas	217	-	-	205	-	-	11
Kentucky	245	-	-	139	-	-	106
Louisiana	407	-	-	151	-	-	256
Maine	44	-	-	31	-	-	13
Maryland	110	-	89	40	-	21	70
Massachusetts	97	-	-	66	-	-	31
Michigan	321	-	-	158	-	-	163
Minnesota	195	-	-	-	-	-	-
Mississippi	126	-	-	127	-	-	-
Missouri	269	-	-	178	-	-	91
Montana	90	-	-	-	-	-	-
Nebraska	153	-	-	146	-	-	7
Nevada	63	-	-	60	-	-	2
New Hampshire	22	-	-	31	-	-	-
New Jersey	106	-	108	50	-	-	56
New Mexico	146	-	-	110	-	-	37
New York	291	-	250	114	-	41	177
North Carolina	243	-	-	107	-	-	136
North Dakota	127	-	-	-	-	-	-
Ohio	392	-	-	152	-	-	240
Oklahoma	312	-	-	245	-	-	67
Oregon	92	-	-	117	-	-	-
Pennsylvania	371	-	276	125	-	94	246
Rhode Island	15	-	-	11	-	-	4
South Carolina	123	-	-	66	-	-	57
South Dakota	39	-	-	36	-	-	3
Tennessee	199	-	-	161	-	-	37
Texas	959	907	798	488	51	160	471
Utah	154	-	-	91	-	-	63
Vermont	12	-	-	-	-	-	-
Virginia	207	-	-	140	-	-	67
Washington	187	-	-	160	-	-	27
West Virginia	176	-	-	92	-	-	84
Wisconsin	162	-	161	109	-	<1	53
Wyoming	155	-	-	116	-	-	39

Source: NERA calculations as explained in text

Note: "-"denotes that all areas of the state would comply with the ozone standard under future baseline conditions according to EPA information in the 2008-2010 ozone analyses and updated emission projections.

"N/A" denotes that U.S. total NO_X emissions for compliance with 84, 75, or 60 ppb are not applicable because not all states have estimated compliance emission levels for each ozone standard.

D. References to the Appendix

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APPENDIX C. ESTIMATES OF STATE-SPECIFIC COMPLIANCE COSTS TO ACHIEVE A 60 PPB OZONE STANDARD

This appendix provides state-specific information on estimated compliance costs for an ozone standard of 60 ppb. As discussed in the report body, we used EPA information on "known" control measures from its 2008-2010 ozone analyses. For states where the "known" control measures would be insufficient to achieve the full necessary NO_X emission reductions for 60 ppb, we supplemented the EPA information with estimates related to coal power plant scrappage and other potential additional control measures as well as with other assumptions to generate a marginal cost curve, i.e., a relationship between marginal cost per ton and the number of tons reduced.

As shown in Appendix B, some states require emission reductions to comply with existing ozone standards of 84 ppb and 75 ppb. We estimate the future compliance costs and economic impacts of *all* ozone NAAQS requirements, including costs attributable to these existing ozone standards.

All cost values in these appendices are shown in 2013 dollars. Compliance costs were developed in 2006 dollars using information from EPA's 2008-2010 analyses, converted to N_{ew}ERA model inputs in 2010 dollars using the U.S. Bureau of Economic Analysis GDP Implicit Price Deflator, and further adjusted to 2013 dollars using the *AEO 2013* GDP Chain-type Price Index.[33]

A. State-Specific Information on "Known" Control Measures

1. EPA Information on "Known" NO_X Controls

In its 2008-2010 ozone analyses, EPA presented state-specific information on "known" NO_X controls from five categories of emission sources: EGUs, non-EGU point sources, area sources, onroad mobile, and nonroad mobile. We developed a comprehensive database of the EPA's information on "known" controls from its 2008-2010 ozone analyses. We removed controls with negative annualized costs or negative NO_X reductions (which are inconsistent with typical emission control analysis) and controls with annualized costs per ton of emission reductions greater than $100,000 (which were also excluded by EPA). We also substituted EGU controls developed using N_{ew}ERA for the EGU controls developed in EPA's analysis (discussed later in this appendix). We then calculated the total annualized costs for "known" control measures in each state, accounting for the possibility of some states not requiring all (or even any) of the "known" control measures from the 2008-2010 ozone analyses based on our calculations of compliance emissions in Appendix B.

Table C-1 provides a national summary of EPA "known" NO_X controls that would be needed for compliance with a new ozone standard of 60 ppb (after removal of controls with negative

[33] The *AEO 2013* price index is available starting in 2010.

reductions, controls with negative costs, and EGU controls). The table shows the specific types of technologies and measures for each emission source category. EPA describes each of these technologies and measures in its 2008 RIA (EPA 2008, Chapter 3 and Appendix 3a). The table also shows the "known" control emission reductions as a percentage of 2018 baseline NO_X emissions for each emission source category (based on the values above in Table B-1). "Known" controls reduce nearly half of baseline NO_X emissions from point sources, but the percentages are much lower for area, onroad mobile, and nonroad mobile sources.

Table C-1. National Summary of EPA "Known" NO_X Controls (tons of reduction)

Point (Non-EGU)	**825,400**
Selective catalytic reduction (SCR) without low-NOx burner (LNB)	466,800
Low-emission combustion (for internal combustion engines)	82,000
Selective catalytic reduction (SCR) and low-NOx burner (LNB)	80,800
Non-selective catalytic reduction (NSCR)	70,300
Selective non-catalytic reduction (SNCR)	61,400
OXY-firing (for glass manufacturers)	33,800
Low-NOx burner (LNB) without selective catalytic reduction (SCR)	20,700
Biosolid injection (for cement kilns)	8,200
Other	1,300
Area	**27,800**
Low-NOx water and space heaters (for commercial buildings)	14,000
Low-NOx burner (LNB)	12,800
Switch to low-sulfur fuel (for residential buildings)	1,000
Onroad Mobile	**256,100**
Retrofit heavy-duty diesel with selective catalytic reduction (SCR)	137,700
Continuous inspection and maintenance	27,800
Eliminate long-duration idling	10,500
Commuting programs	4,400
Low Reid Vapor Pressure	1,000
Unspecified	74,800
Nonroad Mobile	**45,000**
Retrofit heavy-duty diesel with selective catalytic reduction (SCR)	45,000
No Details (Some Omissions in EPA Data for CA and TX)	**130,100**
Total	**1,284,400**

Note: Totals may not equal sum of rows due to independent rounding.
Source: NERA calculations as explained in text

2. State-Specific "Known" NO_X Controls

State-specific information on reductions and annualized costs for "known" controls applied toward a potential standard of 60 ppb are shown in Table C-2. Note that detailed EPA control information was more difficult to assemble for California and Texas, which require controls for compliance with 84 ppb in our analysis (as described above in Appendix B). When detailed source category and cost information for "known" controls were unavailable in these states, we assumed that "known" controls had zero costs.

3. State-Specific "Known" VOC Controls

In its 2008-2010 ozone analyses, EPA includes costs for "known" control measures to reduce emissions of volatile organic compounds (VOCs), particularly from area (non-point) sources but with some additional reductions from other source categories. The NO_X emission reduction requirements that EPA calculated in the 2008-2010 ozone analyses for 60 ppb reflect implementation of VOC controls as well.

We focus on NO_X emissions and emission reductions in our analysis because EPA indicates that NO_X is the critical precursor for ozone formation in most areas of the country, particularly for a tighter new standard of 60 ppb (EPA 2010, pp. S2-3 and S2-14). We apply the VOC costs in our modeling, however, because the information on necessary NO_X emission reductions in the previous EPA analysis assumes implementation of the VOC controls (*i.e.*, necessary NO_X emission reductions in each state for 60 ppb compliance would be different without the VOC controls). Table C-3 shows the estimates of state-level VOC control costs. The VOC control costs are small compared with NO_X control costs (about $800 million in total national annualized costs).

Table C-2. "Known" NO_X Controls by Emission Source Category

	Reductions (tons NOx)					Annualized Costs (million 2013$)				
	Point	Area	Onroad	Nonroad	Total	Point	Area	Onroad	Nonroad	Total
U.S. Total	**825,400**	**27,800**	**256,100**	**45,000**	**1,284,400**	**$3,722**	**$76**	**$850**	**$225**	**$4,873**
Alabama	31,300	900	-	-	32,200	$87	$2	-	-	$89
Arizona	7,000	300	11,300	1,400	20,000	$13	<$1	$31	$7	$52
Arkansas	**8,400**	200	**1,600**	-	**10,300**	**$23**	**<$1**	**$8**	-	**$31**
California	21,700	700	55,900	10,300	161,700 *	$131	$1	$156	$53	$342
Colorado	18,800	500	11,200	900	31,400	$72	<$1	$30	$5	$107
Connecticut	2,500	300	2,100	400	5,400	$7	<$1	$5	$2	$14
Delaware	2,100	100	700	200	3,100	$7	<$1	$2	$1	$10
Florida	15,600	400	-	-	16,000	$46	<$1	-	-	$46
Georgia	16,500	1,500	18,400	1,700	38,000	$45	$3	$70	$8	$126
Idaho	2,400	100	700	-	3,200	$4	<$1	$3	-	$8
Illinois	15,600	200	8,900	3,900	28,500	$120	<$1	$36	$20	$175
Indiana	22,600	700	7,500	2,300	33,000	$93	$1	$32	$11	$138
Iowa	-	-	-	-	-	-	-	-	-	-
Kansas	11,200	-	-	-	11,200	$8	-	-	-	$8
Kentucky	14,400	400	7,500	900	23,200	$71	<$1	$27	$4	$103
Louisiana	126,300	800	3,300	800	131,200	$693	$1	$12	$4	$711
Maine	9,400	<100	1,300	200	10,900	$31	<$1	$5	$1	$37
Maryland	8,300	800	3,200	1,000	13,300	$37	$2	$8	$5	$52
Massachusetts	4,400	1,000	8,500	800	14,700	$15	$2	$23	$4	$43
Michigan	33,800	2,200	8,600	2,200	46,800	$175	$4	$36	$11	$226
Minnesota	-	-	-	-	-	-	-	-	-	-
Mississippi	-	-	-	-	-	-	-	-	-	-
Missouri	17,900	900	11,300	1,400	31,500	$65	$2	$39	$7	$113
Montana	-	-	-	-	-	-	-	-	-	-
Nebraska	3,700	300	-	-	4,100	$7	<$1	-	-	$8
Nevada	400	200	300	-	900	<$1	<$1	-	-	<$1
New Hampshire	-	-	-	-	-	-	-	-	-	-
New Jersey	4,600	1,200	4,900	1,100	11,800	$24	$2	$13	$6	$45
New Mexico	26,000	100	500	-	26,600	$85	<$1	$2	-	$87
New York	18,700	2,600	7,500	2,700	31,500	$70	$25	$25	$13	$134
North Carolina	17,100	300	12,100	1,300	30,800	$54	<$1	$45	$6	$106
North Dakota	-	-	-	-	-	-	-	-	-	-
Ohio	33,900	1,800	15,200	2,700	53,600	$178	$3	$53	$14	$249
Oklahoma	21,900	300	-	-	22,200	$112	<$1	-	-	$113
Oregon	-	-	-	-	-	-	-	-	-	-
Pennsylvania	45,000	1,700	10,900	1,600	59,300	$222	$3	$39	$8	$272
Rhode Island	800	<100	500	<100	1,500	$2	<$1	<$1	<$1	$3
South Carolina	17,300	800	6,600	500	25,300	$64	$2	$28	$3	$97
South Dakota	700	<100	-	-	700	$5	<$1	-	-	$5
Tennessee	25,500	600	2,200	-	28,400	$74	$1	$9	-	$84
Texas	167,800	2,800	13,700	3,500	244,800 *	$888	$6	$41	$17	$952
Utah	6,100	300	5,900	400	12,700	$18	<$1	$15	$2	$35
Vermont	-	-	-	-	-	-	-	-	-	-
Virginia	16,300	2,200	9,600	1,700	29,800	$42	$6	$40	$8	$96
Washington	2,000	<100	-	-	2,100	$24	<$1	-	-	$24
West Virginia	20,200	300	900	-	21,400	$82	<$1	$4	-	$86
Wisconsin	3,500	<100	3,200	800	7,500	$13	<$1	$12	$4	$29
Wyoming	3,600	<100	200	-	3,800	$15	<$1	<$1	-	$16

Source: NERA calculations as explained in text

Note: *California and Texas reduction totals include "known" controls for which we were not able to gather detailed EPA information on emission source category and cost. We assumed that these controls were zero-cost.

Table C-3. EPA VOC Controls by Emission Source Category

	Reductions (tons VOC)					Annualized Costs (million 2013$)				
	Point	Area	Onroad	Nonroad	Total	Point	Area	Onroad	Nonroad	Total
U.S. Total	**4,600**	**187,300**	**95,100**	**11,100**	**298,100**	**$7**	**$588**	**$173**	**$39**	**$807**
Alabama	-	-	-	-	-	-	-	-	-	-
Arizona	<100	5,800	6,400	700	13,000	<$1	$14	$16	$3	$33
Arkansas	-	-	400	-	400	-	-	$1	-	$1
California	300	27,500	<100	<100	27,900	<$1	$55	-	-	$55
Colorado	200	4,700	8,200	400	13,400	<$1	$20	$11	$1	$32
Connecticut	200	13,500	1,800	100	15,700	<$1	$56	<$1	<$1	$58
Delaware	-	-	500	<100	600	-	-	<$1	<$1	<$1
Florida	-	-	-	-	-	-	-	-	-	-
Georgia	-	-	6,800	600	7,400	-	-	$14	$2	$16
Idaho	-	-	-	-	-	-	-	-	-	-
Illinois	-	-	2,100	500	2,500	-	-	$3	-	$3
Indiana	<100	2,700	2,300	300	5,400	<$1	$8	$6	-	$14
Iowa	-	-	-	-	-	-	-	-	-	-
Kansas	-	-	-	-	-	-	-	-	-	-
Kentucky	-	-	2,900	300	3,200	-	-	$6	$1	$7
Louisiana	-	3,900	1,200	400	5,600	-	$17	$5	$2	$24
Maine	-	-	400	200	500	-	-	<$1	<$1	$1
Maryland	100	18,400	2,900	300	21,700	<$1	$68	$2	<$1	$70
Massachusetts	200	3,400	6,700	200	10,500	<$1	$8	$3	<$1	$13
Michigan	600	5,100	3,100	1,400	10,100	<$1	$12	$11	$7	$31
Minnesota	-	-	-	-	-	-	-	-	-	-
Mississippi	-	-	-	-	-	-	-	-	-	-
Missouri	800	1,400	4,600	400	7,200	$1	$3	$7	$2	$12
Montana	-	-	-	-	-	-	-	-	-	-
Nebraska	-	-	-	-	-	-	-	-	-	-
Nevada	-	-	-	-	-	-	-	-	-	-
New Hampshire	-	-	-	-	-	-	-	-	-	-
New Jersey	100	29,300	3,600	400	33,400	<$1	$117	$2	$1	$121
New Mexico	-	-	200	-	200	-	-	<$1	-	<$1
New York	500	15,000	6,200	1,000	22,600	<$1	$26	$9	$4	$39
North Carolina	-	-	4,500	500	5,000	-	-	$12	$2	$13
North Dakota	-	-	-	-	-	-	-	-	-	-
Ohio	200	16,700	8,300	1,000	26,300	<$1	$68	$20	$4	$93
Oklahoma	-	-	-	-	-	-	-	-	-	-
Oregon	-	-	-	-	-	-	-	-	-	-
Pennsylvania	400	23,000	5,600	700	29,800	<$1	$79	$13	$3	$96
Rhode Island	-	-	700	<100	700	-	-	<$1	<$1	<$1
South Carolina	-	-	2,300	300	2,600	-	-	$8	$1	$10
South Dakota	-	-	-	-	-	-	-	-	-	-
Tennessee	-	-	700	-	700	-	-	$2	-	$2
Texas	400	8,800	1,900	500	11,600	<$1	$21	-	-	$21
Utah	<100	1,000	4,300	200	5,600	<$1	$2	$4	<$1	$7
Vermont	-	-	-	-	-	-	-	-	-	-
Virginia	<100	1,600	5,700	500	7,900	<$1	$4	$11	$2	$17
Washington	-	-	-	-	-	-	-	-	-	-
West Virginia	-	-	400	-	400	-	-	$2	-	$2
Wisconsin	400	5,300	300	100	6,100	<$1	$11	<$1	-	$12
Wyoming	-	-	<100	-	<100	-	-	<$1	-	<$1

Source: NERA calculations as explained in text

B. State-Specific Information on EGU Controls

1. Development of EGU Controls in N_{ew}ERA

We used the electricity module of the N_{ew}ERA model to estimate the net changes in NO_X emissions and electricity system costs for potential EGU controls for 60 ppb compliance. In particular, in N_{ew}ERA, we did not allow coal units to operate (and emit NO_X) in states requiring emission reductions from "unknown" controls, and we developed state-specific estimates of the net NO_X reduction from EGU source categories, net costs, and costs per ton of NO_X removed.

a. Assumptions

We use the N_{ew}ERA model to estimate the potential NO_X reductions from scrapping coal-fired generation and the estimated cost per ton associated with this action. As part of the model runs we assume that there would not be any NO_X emissions from any coal-fired generation in the relevant states. We performed separate analyses for each of the 44 U.S. states that have existing coal-fired generation. We compared these results to a baseline run without coal-specific NO_X restrictions.

For each state, the coal-specific NO_X restriction was imposed in 2026 and future years. We selected 2026 for all states for purposes of consistency, even though some states would make reductions prior to 2026, and others would not make reductions until after 2026. We do not believe that changing the year would significantly change the estimated costs per ton removed. For each state with a coal-specific NO_X restriction, we took the difference between total 2026 U.S. electric sector NO_X emissions from the state-specific model run and the same emissions from the baseline run without coal-specific NO_X restrictions. This provided us with the net NO_X emission reductions (accounting for the reductions in NO_X from coal units in the state of interest, increases in NO_X emissions from natural gas-fired generation from the state of interest and surrounding states, and increases in NO_X emissions from coal-fired generation from surrounding states). This approach may understate or overstate the NO_X emission reductions because we have effectively assumed that the state of interest would be the only state to impose the NO_X restrictions on its coal-fired fleet. NO_X emission reductions would be understated if the lost generation from in-state coal generators could not be replaced by coal-fired generation from surrounding states. NO_X emission reductions would be overstated if the natural gas-fired generation in surrounding states were not available to generate more because it was already generating to replace lost coal-fired generation from within its own state as might happen if that state were to also impose a coal-specific NO_X restriction.

To calculate the increased costs resulting from the coal-specific NO_X restriction, we took the difference between total U.S. electric sector costs in the state-specific run and the baseline run without coal-specific NO_X restrictions in 2026. All capital costs were annualized for purposes of this cost comparison. These costs are mostly sourced from *AEO 2013* and *AEO 2014*, including capital and operating costs of new natural gas-fired generators and other types of generators

(*AEO 2013*), natural gas prices (*AEO 2014*), and electricity demand (*AEO 2014*). The increases in costs therefore reflect the following: changes in fuel costs resulting from operating higher cost generating units (either natural gas-fired, coal-fired, or other dispatchable resources), changes in O&M costs from the change in dispatch, and annualized costs associated with building new generating resources (if necessary). The costs are likely understated for some of the same reasons that emission reductions might be overstated; but also because if many states were to impose coal NO_X limits, then electric sector demand for natural gas would likely increase, thereby increasing natural gas prices (as shown in our main results). For this analysis of potential EGU controls, natural gas prices were assumed to be unchanged.

b. Cost per Ton of NO_X Reduced by State

Table C-4 shows the cost per ton of NO_X removed by state based upon the assumption that coal-fired generation from existing EGU sources would not occur in 2026 and beyond for the relevant states.

Oregon only has one existing coal plant, Boardman, and that plant is scheduled to retire prior to 2026, thus there are not any changes in NO_X emissions or costs. South Dakota and Georgia show increases in NO_X emissions, which happens because lower NO_X emitting coal-fired generation in those states is replaced by imports of electricity, which have higher NO_X emissions per unit of power. Finally, Connecticut shows a small decrease in costs in 2026 along with a decrease in NO_X emissions. While costs decline in 2026, costs increase over the entire model horizon, so this is just an intertemporal impact.

2. Combination of EGU Controls with EPA "Known" Controls

For states where coal power plant scrappage and replacement would cost less than $30,000 per ton, we included this measure with the EPA "known" controls in the marginal cost curve. For states where it would cost more than $30,000 per ton, we included this measure in the segment of the marginal cost curve reflecting "unknown" controls, as explained below.

C. State-Specific Information on "Unknown" Controls

1. Necessary NO_X Emission Reductions from Additional Controls

We estimated the necessary NO_X emission reductions from additional controls for each state by calculating the gap between each state's necessary total NO_X emission reductions for 60 ppb and reductions that would be achieved from "known" controls and N_{ew}ERA EGU controls costing less than $30,000 per ton. These additional controls represent either EGU controls costing more than $30,000 per ton or "unknown" controls for which we do not have any detailed information. Table C-5 shows the calculation of reductions from additional controls for each state.

Table C-4. Summary of N_{ew}ERA EGU Control Modeling in 2026

	Average Cost per Ton Removed (2013$)
Alabama	$18,241
Arkansas	$10,491
Arizona	$18,411
California	$3,970
Colorado	$25,024
Connecticut	$0
Delaware	$184,330
Florida	$57,770
Georgia	N/A
Iowa	$24,593
Illinois	$68,039
Indiana	$29,061
Kansas	$84,506
Kentucky	$25,470
Louisiana	$17,671
Massachusetts	$31,329
Maryland	$15,024
Michigan	$31,528
Minnesota	$37,828
Missouri	$46,412
Mississippi	$19,888
Montana	$19,104
North Carolina	$28,351
North Dakota	$14,114
Nebraska	$28,692
New Hampshire	$20,177
New Jersey	$78,909
New Mexico	$47,929
Nevada	$2,757
New York	$10,482
Ohio	$30,612
Oklahoma	$9,741
Oregon	N/A
Pennsylvania	$43,014
South Carolina	$5,933
South Dakota	N/A
Tennessee	$47,808
Texas	$63,272
Utah	$8,536
Virginia	$7,393
Washington	$4,307
Wisconsin	$70,241
West Virginia	$119,580
Wyoming	$26,874

Note: "N/A" denotes that the state would not achieve net NO_X reductions from coal power plant scrappage. States are omitted if they do not have any coal power plants.
Source: NERA calculations as explained in text

Table C-5. Additional Reductions Required After EPA "Known" and N_{ew}ERA EGU Controls Costing Less than $30,000 per Ton (tons of NO_X)

	Updated Baseline Emission Projections	Reductions for 60ppb (from Updated Baseline)	EPA "Known" Reductions	Coal Power Plant Scrappage Reductions (<$30,000/ton)	Additional Reductions Needed
U.S. Total	**9,704,600**	**3,866,100**	**1,284,400**	**698,200**	**1,883,500**
Alabama	262,300	74,700	32,200	42,500	-
Arizona	152,300	43,200	20,000	23,100	-
Arkansas	138,300	27,300	10,300	17,000	-
California	553,400	497,300	161,700	-	335,500
Colorado	203,300	81,000	31,400	32,700	16,900
Connecticut	48,500	30,900	5,400	600	24,900
Delaware	19,700	6,800	3,100	-	3,700
Florida	358,000	29,700	16,000	-	13,700
Georgia	244,600	70,400	38,000	14,400	18,000
Idaho	57,700	4,600	3,200	-	1,400
Illinois	347,500	193,700	28,500	-	165,200
Indiana	319,100	193,300	33,000	106,200	54,100
Iowa	173,900	-	-	-	-
Kansas	216,500	11,200	11,200	-	-
Kentucky	245,000	106,200	23,200	82,000	1,000
Louisiana	406,900	256,400	131,200	11,200	113,900
Maine	43,600	12,700	10,900	-	1,800
Maryland	110,000	69,800	13,300	11,000	45,500
Massachusetts	96,900	30,900	14,700	3,900	12,400
Michigan	320,500	162,800	46,800	55,700	60,300
Minnesota	194,900	-	-	-	-
Mississippi	126,400	-	-	-	-
Missouri	268,500	90,600	31,500	-	59,200
Montana	90,400	-	-	-	-
Nebraska	153,200	6,800	4,100	2,700	-
Nevada	62,500	2,300	900	1,400	-
New Hampshire	22,200	-	-	-	-
New Jersey	106,300	56,000	11,800	-	44,200
New Mexico	146,100	36,600	26,600	-	9,900
New York	291,400	177,400	31,500	5,900	140,000
North Carolina	243,400	136,100	30,800	36,900	68,500
North Dakota	127,200	-	-	-	-
Ohio	392,300	240,000	53,600	86,600	99,800
Oklahoma	311,900	67,100	22,200	44,900	-
Oregon	91,600	-	-	-	-
Pennsylvania	370,800	245,600	59,300	-	186,200
Rhode Island	14,900	3,600	1,500	-	2,100
South Carolina	123,300	57,100	25,300	9,800	22,000
South Dakota	39,200	3,400	700	-	2,700
Tennessee	198,700	37,300	28,400	-	8,900
Texas	958,600	471,000	244,800	-	226,200
Utah	153,900	62,800	12,700	50,100	-
Vermont	12,300	-	-	-	-
Virginia	206,700	67,100	29,800	19,200	18,200
Washington	187,000	27,300	2,100	5,400	19,800
West Virginia	176,200	83,900	21,400	-	62,500
Wisconsin	161,600	52,600	7,500	-	45,000
Wyoming	155,100	38,600	3,800	34,800	-

Source: NERA calculations as explained in text

2. Estimating the Cost of Additional Controls

a. Overview of the NERA Marginal Cost Curve

As mentioned in the report body, we performed detailed analyses of residual emissions from each source category in each state assuming implementation of all the EPA "known" controls to develop the bases for updated estimates of potential additional control options and their costs. We developed illustrative extensions of each state's marginal cost curve for "unknown" controls (as needed) using as an anchor point a particular potential additional control at an estimated cost per ton and an estimated placement along the horizontal axis reflecting cumulative NO_X emission reductions up to that point. In particular, our anchor point for the segment of each state's marginal cost curve for "unknown" controls reflects scrappage of older high-emission-rate passenger cars and light duty trucks and replacement with new low-emission-rate vehicles.

To estimate the cost per ton of this anchor point, we adapted the methodology used in Knittel (2009) to estimate the cost-effectiveness of the Consumer Assistance to Recycle and Save Act (CARS or "Cash for Clunkers"). The Knittel (2009) model compares the average rebate paid for scrapping an existing vehicle and purchasing a new one through the CARS program (the cost) with an estimate of emission reductions achieved by the vehicle trade-in. We use a similar framework with different assumptions to estimate the cost per ton of reducing NO_X emissions through a vehicle scrappage program in an illustrative *future* year of 2020 (the first attainment year in our analysis). We assumed the target age of vehicles scrapped in the future program would be 13 years with an expected remaining useful life of about 45,000 miles based on the survival probabilities and average VMT by vehicle vintage shown in Knittel (2009), Table 2.[34] We assumed the rebate (the cost per vehicle) would be about $4,200 in 2009 dollars, the same as the average rebate paid through the CARS program.[35] We annualized the rebate over a remaining useful vehicle life of about 4 years at a 5% discount rate.[36]

Newer vehicles are required to meet more stringent NO_X emission standards, so NO_X emissions are reduced as older vehicles are scrapped and replaced with new vehicles. The emission rate for scrapped vehicles was estimated as the average of NO_X emission rate standards for new cars and light trucks in 2007, the relevant emission rate standards for vehicles that will be 13 years old in

[34] Survival probabilities are from the NHTSA 2006 Vehicle Survivability and Travel Mileage Schedule. We use the average of car and light truck expected VMT remaining, weighted by the 2020 car and light truck stock projected in *AEO 2014* and conditional on survival to age 13 as in Knittel (2009).

[35] The average rebate assumed in Knittel (2009) is $4,200, and the average age of vehicles scrapped through the CARS program was 14 years (NHTSA, 2009 p. 21). We assume a program aiming to scrap vehicles averaging less than 13 years of age would require a somewhat higher average rebate; for programs targeting older vehicles, we select a rebate by multiplying the *AEO 2014* projected new light duty vehicle price for 2020 by the target vehicle's share of remaining lifetime VMT, scaled by a calibration factor to return a rebate of $4,200 for a program targeting 13-year-old vehicles (as observed in the CARS program).

[36] Annualization years were based on vehicle miles remaining (described above) and an assumed average of 12,000 miles per year from Knittel (2009).

2020.[37] The resulting emission rate for scrapped vehicles was 0.19 g/mile. Finally, we assumed new replacement vehicles would have an emission rate of 0.03 g/mile to meet EPA Tier 3 emission rate standards and calculated annual NO_X emission reductions per vehicle of 0.0022 tons. Comparing the annualized rebate cost to the annual emission reductions gave a cost per ton of about $540,000.

Using VMT by age information shown in Knittel (2009) and the 1999 age distribution of the car and light truck stock from MOVES2010 documentation (EPA 2010a), we estimate that scrapping light-duty vehicles 13 years old and older would eliminate about 50% of light-duty vehicle emissions. As explained above, the marginal cost of scrapping enough of the fleet to achieve a 50% NO_X emissions reduction from passenger cars and light duty trucks would be approximately $500,000 per ton, and we assume that this option would be undertaken at the part of each state's marginal cost curve corresponding to a 75% reduction from the baseline NO_X emission level. This calculation is based on estimates for NO_X emissions rates by age of passenger cars and light duty trucks, survival rates and VMT for vehicles, estimated rebates that would need to be paid to vehicles owners to scrap their vehicles, and the remaining useful life of a vehicle when it would be scrapped.

This anchor point, in combination with cost-per-ton information for "known" controls (including coal scrappage if less than $30,000/ton in the state), determines the slope of the "unknown" segment of each state's marginal cost curve. We estimate total annualized costs for "unknown" controls for each state using this slope and the necessary remaining tons of NO_X emission reductions after implementation of "known" controls (including coal scrappage if less than $30,000/ton in the state). Section II of the report illustrates the nature of the marginal cost curve we presume, showing the anchor points and the segment of the curve representing costs for "unknown" control measures.

The increasing marginal cost curve can be motivated by considering the steps involved in expanding a program to scrap existing passenger cars and light duty trucks. Using the same methodology described above, we estimate that a vehicle scrappage program targeting older vehicles than our example above could achieve a 10% reduction in light-duty vehicle emissions at a marginal cost of about $120,000 per ton. In contrast, newer vehicles have a longer useful life and would thus require a larger rebate to incentivize their owners to scrap and replace them with lower-emitting vehicles. Scrapping newer vehicles also reduces emissions less than scrapping older, higher-emitting vehicles, so the marginal cost of reducing vehicle emissions would thus increase as a program moves from scrapping older vehicles to scrapping a larger share of the vehicle fleet (including newer vehicles).

[37] We estimate vehicle NO_X emission rates for each vehicle vintage by linearly interpolating between historical federal NO_X emission standards for new vehicles.

b. Implications for Other Source Categories

We believe that other source categories have a rising marginal cost curve similar to that of passenger cars and light duty trucks. We have estimates of the emissions that remain in these source categories after accounting for "known" controls and can infer an approximate minimum cost for these reductions (otherwise these reductions would have been included in "known" controls). Further, the remaining NO_X emissions from many of these other source categories reflect different vintages of equipment presumably with improving NO_X emissions rates over time as efficiencies of the equipment have improved, much like that of passenger cars and light duty trucks. Thus, we think it is reasonable that the cost of reductions in these sectors would conform to a marginal cost curve similar to the form for passenger cars and light duty trucks.

3. Reductions and Costs from Additional Controls by State

Table C-6 shows emission reductions from additional controls beyond "known" controls and EGU controls costing more than $30,000 per ton. The costs of "unknown" controls were estimated at the state level using the cost curve methodology discussed above. The final compliance costs for all EGU controls were determined endogenously in the N_{ew}ERA model and cannot be isolated from other electricity sector impacts of the ozone standards.

We allocated total estimated costs for "unknown" controls (shown in Table C-6) to the four emission source categories other than EGU: (1) point; (2) area; (3) onroad mobile; and (4) nonroad mobile. This allocation was based on the state-level emissions remaining in each source category after applying the EPA "known" controls. For example, if 30% of a state's non-EGU emissions after implementing "known" controls were in the nonroad emission source category, we placed 30% of that state's "unknown" control costs in the nonroad source category. Table C7 shows the share of "unknown" control costs allocated to each emission source category by state.

D. Total State-Specific Annualized Compliance Costs

Table C-8 summarizes our total non-EGU annualized compliance cost estimates by state, separated between "known" and "unknown" control measures. Note that annualized costs for "known" control measures include costs for VOC controls in addition to costs for NO_X controls. Final compliance costs for all EGU controls were determined endogenously in the N_{ew}ERA model and are not included in Table C-8.

Table C-6. Costs of Additional Controls by State

	Reductions from Additional Controls (tons NO_x)			Costs of Additional Controls (million 2013$)	
	Coal Power Plant Scrappage (>$30,000/ton)	"Unknown"	Total	Coal Power Plant Scrappage (>$30,000/ton)	"Unknown"
U.S. Total	**295,200**	**1,588,300**	**1,883,500**	**Modeled**	**$342,159**
Alabama	-	-	-	-	-
Arizona	-	-	-	-	-
Arkansas	-	-	-	-	-
California	-	335,500	335,500	-	$128,751
Colorado	-	16,900	16,900	-	$1,309
Connecticut	-	24,900	24,900	-	$6,112
Delaware	-	3,700	3,700	-	$395
Florida	-	13,700	13,700	-	$466
Georgia	-	18,000	18,000	-	$1,065
Idaho	-	1,400	1,400	-	$23
Illinois	48,200	117,000	165,200	-	$18,973
Indiana	-	54,100	54,100	-	$9,475
Iowa	-	-	-	-	-
Kansas	-	-	-	-	-
Kentucky	-	1,000	1,000	-	$28
Louisiana	-	113,900	113,900	-	$24,527
Maine	-	1,800	1,800	-	$81
Maryland	-	45,500	45,500	-	$10,710
Massachusetts	-	12,400	12,400	-	$1,153
Michigan	-	60,300	60,300	-	$9,012
Minnesota	-	-	-	-	-
Mississippi	-	-	-	-	-
Missouri	38,600	20,600	59,200	-	$1,132
Montana	-	-	-	-	-
Nebraska	-	-	-	-	-
Nevada	-	-	-	-	-
New Hampshire	-	-	-	-	-
New Jersey	2,600	41,600	44,200	-	$7,876
New Mexico	9,900	-	9,900	-	-
New York	-	140,000	140,000	-	$32,816
North Carolina	-	68,500	68,500	-	$12,983
North Dakota	-	-	-	-	-
Ohio	-	99,800	99,800	-	$20,485
Oklahoma	-	-	-	-	-
Oregon	-	-	-	-	-
Pennsylvania	43,800	142,500	186,200	-	$28,791
Rhode Island	-	2,100	2,100	-	$170
South Carolina	-	22,000	22,000	-	$2,778
South Dakota	-	2,700	2,700	-	$107
Tennessee	4,800	4,100	8,900	-	$116
Texas	72,000	154,200	226,200	-	$17,070
Utah	-	-	-	-	-
Vermont	-	-	-	-	-
Virginia	-	18,200	18,200	-	$1,303
Washington	-	19,800	19,800	-	$1,185
West Virginia	62,500	-	62,500	-	-
Wisconsin	12,800	32,300	45,000	-	$3,264
Wyoming	-	-	-	-	-

Note: Coal power plant scrappage compliance costs are determined in the N_{ew}ERA model.
Source: NERA calculations as explained in text

Table C-7. Cost Allocation for "Unknown" Controls to Non-EGU Source Categories

	Non-EGU "Unknown" Control Annualized Costs (million 2013$)	Share of Non-EGU Emissions Remaining After "Known" Controls			
		Point	Area	Onroad	Nonroad
U.S. Total	$342,159	14%	24%	34%	29%
Alabama	-	-	-	-	-
Arizona	-	-	-	-	-
Arkansas	-	-	-	-	-
California	$128,751	11%	14%	42%	33%
Colorado	$1,309	24%	33%	22%	21%
Connecticut	$6,112	6%	29%	31%	34%
Delaware	$395	0%	15%	36%	49%
Florida	$466	12%	8%	45%	35%
Georgia	$1,065	18%	10%	48%	24%
Idaho	$23	16%	12%	44%	29%
Illinois	$18,973	20%	22%	24%	34%
Indiana	$9,475	25%	15%	34%	26%
Iowa	-	-	-	-	-
Kansas	-	-	-	-	-
Kentucky	$28	13%	33%	30%	24%
Louisiana	$24,527	0%	37%	16%	47%
Maine	$81	12%	34%	26%	28%
Maryland	$10,710	10%	15%	47%	28%
Massachusetts	$1,153	13%	28%	25%	34%
Michigan	$9,012	12%	30%	36%	22%
Minnesota	-	-	-	-	-
Mississippi	-	-	-	-	-
Missouri	$1,132	6%	9%	57%	28%
Montana	-	-	-	-	-
Nebraska	-	-	-	-	-
Nevada	-	-	-	-	-
New Hampshire	-	-	-	-	-
New Jersey	$7,876	9%	25%	29%	37%
New Mexico	-	-	-	-	-
New York	$32,816	9%	30%	34%	27%
North Carolina	$12,983	12%	15%	51%	23%
North Dakota	-	-	-	-	-
Ohio	$20,485	10%	16%	49%	25%
Oklahoma	-	-	-	-	-
Oregon	-	-	-	-	-
Pennsylvania	$28,791	7%	40%	34%	19%
Rhode Island	$170	4%	43%	28%	26%
South Carolina	$2,778	11%	13%	47%	29%
South Dakota	$107	7%	20%	24%	49%
Tennessee	$116	10%	21%	45%	25%
Texas	$17,070	7%	40%	31%	22%
Utah	-	-	-	-	-
Vermont	-	-	-	-	-
Virginia	$1,303	14%	19%	38%	29%
Washington	$1,185	12%	5%	48%	34%
West Virginia	-	-	-	-	-
Wisconsin	$3,264	20%	17%	36%	27%
Wyoming	-	-	-	-	-

Note: "-" indicates that there were no non-EGU "unknown" control costs in the state, and therefore no need to estimate emissions remaining in each source category after "known" controls.
Source: NERA calculations as explained in text

Table C-8. Annualized Compliance Costs of a 60 ppb Standard by Control Type and State (million 2013$)

	"Known" Control Costs	"Unknown" Control Costs	Total Non-EGU Control Costs
U.S. Total	**$5,703**	**$342,159**	**$347,862**
Alabama	$89	-	$89
Arizona	$85	-	$85
Arkansas	$32	-	$32
California	$397	$128,751	$129,148
Colorado	$139	$1,309	$1,448
Connecticut	$72	$6,112	$6,184
Delaware	$10	$395	$406
Florida	$46	$466	$513
Georgia	$142	$1,065	$1,207
Idaho	$9	$23	$32
Illinois	$178	$18,973	$19,151
Indiana	$152	$9,475	$9,628
Iowa	-	-	-
Kansas	$8	-	$8
Kentucky	$111	$28	$139
Louisiana	$735	$24,527	$25,262
Maine	$39	$81	$120
Maryland	$122	$10,710	$10,833
Massachusetts	$56	$1,153	$1,209
Michigan	$256	$9,012	$9,269
Minnesota	-	-	-
Mississippi	-	-	-
Missouri	$126	$1,132	$1,258
Montana	-	-	-
Nebraska	$8	-	$8
Nevada	$23	-	$23
New Hampshire	-	-	-
New Jersey	$166	$7,876	$8,042
New Mexico	$88	-	$88
New York	$173	$32,816	$32,989
North Carolina	$119	$12,983	$13,102
North Dakota	-	-	-
Ohio	$342	$20,485	$20,827
Oklahoma	$113	-	$113
Oregon	-	-	-
Pennsylvania	$369	$28,791	$29,160
Rhode Island	$4	$170	$174
South Carolina	$106	$2,778	$2,885
South Dakota	$5	$107	$112
Tennessee	$86	$116	$202
Texas	$973	$17,070	$18,043
Utah	$42	-	$42
Vermont	-	-	-
Virginia	$113	$1,303	$1,417
Washington	$24	$1,185	$1,209
West Virginia	$88	-	$88
Wisconsin	$41	$3,264	$3,305
Wyoming	$16	-	$16

Note: "-" indicates that there were no non-EGU "unknown" control costs in the state, and therefore no need to estimate emissions remaining in each source category after "known" controls.
Source: NERA calculations as explained in text

E. EPA Compliance Cost Estimates

This section discusses issues related to EPA's estimates of total annualized costs for 60 ppb in its 2010 supplemental ozone analysis. In this section, we present the estimates, note important details regarding their calculations and scope, and summarize key differences from the cost estimates that we developed for this analysis. As the final section of this section, we also discuss implications of these issues for the forthcoming EPA ozone analysis.

1. EPA Total Cost Estimates

EPA provided information on the necessary emission reductions and costs for 60 ppb compliance in its 2010 supplemental ozone analysis (EPA 2010b). EPA performed its analysis for a single future year: 2020. Its calculations of necessary emission reductions and costs for 60 ppb reflect its projections of baseline emissions and baseline air quality in that future year.

EPA used two approaches to estimate the costs of "unknown" controls in its 2008-2010 ozone analyses, and it developed estimates using various parameters for each of the two approaches. In the "fixed" approach, EPA assumed that all "unknown" controls would have a constant cost per ton, and it developed estimates using $10,000, $15,000, and $20,000 per ton as parameters. In the "hybrid" approach, EPA assumed that "unknown" controls would begin at $15,000 per ton in each area and would gradually increase in marginal cost based on possible slope parameters. EPA developed estimates for the "hybrid" approach using 0.12, 0.24, and 0.48 as the slope parameters (EPA 2008, pp. 5-10 to 5-22; EPA 2010b, pp. S2-17 to S2-18).[38]

Table C-9 reproduces the total annualized cost estimates for 60 ppb from EPA (2010b). As shown in the table, EPA estimated that "known" controls for 60 ppb compliance in 2020 would have a total annualized cost of $4.5 billion (in 2006 dollars). The cost estimates for "unknown" controls reflect the two approaches with their middle parameters (*i.e.*, the "fixed" approach assuming $15,000 per ton and the "hybrid" approach assuming 0.24 as the slope parameter). Thus, the total cost for 60 ppb in the EPA 2010 analysis was about $52 billion for the "fixed" approach with this assumed cost per ton and about $90 billion for the "hybrid" approach with this assumed slope parameter.

[38] NERA's approach for estimating the costs of "unknown" controls is similar to EPA's "hybrid" approach in using an upward-sloped marginal cost curve, as described in Appendix C.

Table C-9. EPA Total Annualized Cost Estimates for 60 ppb from 2010 RIA (billion 2006$)

"Known" Controls	$4.5	
"Unknown" Controls	"Fixed": $47	"Hybrid": $85
Total	"Fixed": $52	"Hybrid": $90

Note: Costs reflect compliance actions for 2020.
"Fixed" approach for costs of "unknown" controls reflects assumption of $15,000 per ton.
"Hybrid" approach for costs of "unknown" controls reflects assumption of 0.24 slope parameter.
Source: EPA (2010b), Table S2.9 (p. S2-19)

The following points should be noted regarding the EPA cost estimates from the 2010 analysis.

- *Annualized costs for 2020.* As noted above, EPA only evaluated a single future year (2020) for its previous ozone analysis. EPA did not address staggered implementation of compliance deadlines for nonattainment areas in its analysis. Its cost estimates reflect annualized costs that combine capital expenditures (converted from lump sums to annualized values) and operating expenditures (incurred on an annual basis). It did not calculate a present value based on its annualized cost estimates.

- *Ranges of cost estimates.* As noted above, EPA used various approaches and various parameters to estimate the costs of "unknown" controls for 60 ppb compliance. Its main table of results (reproduced above) uses the middle parameters for the two approaches. Results using other parameters are shown in Docket File 399 (EPA 2010c). This file shows that for the "fixed" approach, "unknown" controls would cost $32 billion (in 2006 dollars) using $10,000 per ton and $63 billion using $20,000 per ton. It also shows that for the "hybrid" approach, "unknown" controls would cost $66 billion using 0.12 as the slope parameter and $123 billion using 0.48 as the slope parameter.

- *Exclusion of Southern and Central California.* EPA did not include ozone compliance costs for Southern and Central California in its main cost estimates, because these areas would have extra time (beyond 2020) to improve their air quality (EPA 2008, pp. 4-3, 5-10, and 7b-1 to 7b-14; EPA 2010b, p. S2-19).

- *Error in "Hybrid" Approach Calculations.* The spreadsheet that EPA used to calculate the costs for "unknown" controls with the "hybrid" approach contains an error. Information for Cleveland, Mississippi was erroneously applied to Cleveland, Ohio, leading to incorrect cost estimates for both areas. As a result of this error, cost estimates for 60 ppb based on the "hybrid" approach were too high by about $8 billion (in 2006 dollars) using the low slope parameter of 0.12, by about $15 billion using the middle slope parameter of 0.24, and by about $30 billion using the high slope parameter of 0.48. While EPA was made aware of these errors, it was instructed to terminate the ozone reconsideration in September 2011 (OMB 2011), and as a result corrected cost estimates were never issued.

2. Differences from NERA Cost Estimates

The EPA total cost estimates that are summarized above differ from cost estimates in this study because NERA has undertaken an improved, more evidence-based approach for developing reasoned estimates of the costs for the two-thirds of the controls that EPA treated as "unknown" in its prior analysis. EPA made estimates of the cost per ton for those two-thirds of the overall controls without any reference to or assessment of the actual types of controls that would logically have to be undertaken to eliminate those additional tons. Our assessment, having used available information to estimate the range of costs per ton that would be associated with those remaining types of emissions sources, indicates that EPA's earlier extrapolation assumptions were probably too low. There are a number of other reasons why NERA's cost estimates differ from those in the earlier EPA RIA:

- *Annualized vs. annual costs and present values.* As discussed earlier in this appendix, NERA used annualized cost information for "known" controls from the EPA 2008-2010 ozone analysis and supplemented it with modeling of coal-fired generation unit scrappage and other potential additional controls to estimate the costs of "unknown" controls, which we also calculated on an annualized basis. As discussed in Appendix D, the costs for each state were divided between capital and operating expenditures and were entered into various years in N_{ew}ERA based on potential state designations and compliance deadlines. The report body also shows the estimated costs as a present value. Thus, the NERA cost information accounts for staggered implementation of the ozone standard across states and includes a present value, in contrast to the EPA analysis.

- *2020 vs. staggered implementation.* As discussed above, EPA used baseline emissions and air quality projections for 2020 to estimate necessary emission reductions and associated costs for 60 ppb compliance in that year. NERA incorporated staggered implementation of the ozone standard into the modeling by specifying potential designations and compliance deadlines for each state and developing cost estimates by year based on those designations.

- *Exclusion vs. inclusion of Southern and Central California.* In contrast to the EPA analysis, NERA included all parts of the country (with available information) in the cost estimates and economic impact modeling. As with other areas, NERA specified a potential ozone nonattainment designation and compliance deadline for California for each relevant level (84, 75, and 60 ppb). Our analyses related to California are described in Appendices B, C, and D.

- *All sectors vs. separate modeling of electricity sector costs.* The EPA cost estimates reflect emission reductions in all sectors, including the electricity sector. In contrast, NERA used the electricity sector module of the N_{ew}ERA modeling system to analyze emission reductions in that sector, and compliance costs for the electricity sector are not included in NERA's cost estimates.

- *2006 vs. 2013 dollars.* EPA presented its cost estimates in the 2008-2010 ozone analyses in 2006 dollars. NERA presents costs in 2013 dollars.

For these reasons, the EPA and NERA cost estimates cannot be compared on an "apples-to-apples" basis.

3. Implications for Forthcoming EPA Analysis

This discussion of EPA's total cost estimates from its 2010 analysis and differences with NERA's analyses has several implications for EPA's forthcoming ozone analysis.

- *Staggered implementation.* Analysis for a single future year has limited usefulness when different areas of the country would have different compliance timelines (and may also still need to comply with previously promulgated standards, such as the 1997 ozone standard of 84 ppb and the 2008 ozone standard of 75 ppb). EPA should incorporate staggered implementation into its forthcoming ozone analysis for a more realistic treatment of timing issues. In addition, expanding the analysis beyond a "snapshot" year (*e.g.*, 2020) would allow EPA to include Southern and Central California in its compliance cost estimates.

- *Identification of additional controls and cost estimation for any "unknown" controls.* In EPA's 2010 analysis of 60 ppb, "known" controls achieved less than one-half of the necessary total emission reductions in many areas of the country, and the total costs for "unknown" controls were much larger than the total costs for "known" controls, using either the "fixed" or "hybrid" approach for "unknown" control cost estimation and using any of the parameters. Moreover, EPA's cost estimates for "unknown" controls vary widely, from about $32 billion (in 2006 dollars) using the "fixed" approach and low parameter of $10,000 per ton to about $123 billion using the "hybrid" approach and high slope parameter of 0.48 (or about $93 billion after correcting the calculation error). To reduce the reliance on "unknown" controls for this regulation and reduce the uncertainty regarding its costs, EPA should dedicate more resources to identifying additional potential control measures and developing more precise estimates of the cost of any remaining "unknown" controls.

F. References to the Appendix

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APPENDIX D. ESTIMATES OF STATE-SPECIFIC COMPLIANCE COST INPUTS TO N_{EW}ERA TO ACHIEVE A 60 PPB OZONE STANDARD

A. Allocation of Costs to N_{ew}ERA Sectors

As described in Appendix C, we developed state-level compliance costs by control type ("known," coal scrappage, and "unknown") and by emission source category (point, area, onroad, nonroad, and EGU).[39] In order to model the economic impacts of these compliance costs, we then divided costs in each emission source category among economic sectors used in the N_{ew}ERA model. This section describes those sector allocations.

In general, the same methodology was used to allocate "known" and "unknown" control costs to N_{ew}ERA sectors. Table D-1 and Table D-2 show respectively the resulting national-level shares of "known" and "unknown" control costs by source category and N_{ew}ERA sector.[40] The national sector composition of each source category is different in the two tables; this primarily reflects state-level differences in the source categories of "unknown" control costs and state-specific sector allocations.

Table D-3 shows the resulting state-level annualized cost inputs by N_{ew}ERA sector based on both "known" and "unknown" controls. These costs reflect the many different types of controls that could be required for the various emission sources within each N_{ew}ERA sector to achieve a new ozone NAAQS of 60 ppb. For example:

- Services. Businesses in the services sector could be required to install lower-emitting equipment for space heating, air conditioning and water heaters. Service sector businesses also operate substantial numbers of transportation vehicles, and the older portions of those fleets face the need for early scrappage, which accounts for a fair share of the total costs. Similarly, but a smaller part of the total cost, service sector businesses use landscaping equipment that would also need to be replaced with lower-emitting versions.

- Commercial transportation except trucking. This category includes locomotives, airplanes, river and ocean-going vessels in port, and off-road commercial equipment (bulldozers, dump trucks, etc.). A small fraction of the cost in this category comes from retrofitting NO_X reduction equipment onto existing engines. The larger part of the costs comes from assumed replacement of current equipment with lower-emitting versions,

[39] EPA's 2008-2010 analyses provide the emission source category of each "known" control. As discussed in Appendix C, we divided "unknown" control costs among source categories using state-level emissions remaining after applying any "known" controls in each state. Coal scrappage is in the EGU emission source category.

[40] Sector allocations were developed at the state level and therefore vary by state.

including electrification of locomotives and auxiliary ship engines (for use while at dock).

- Household transportation. This sector primarily reflects personal on-road vehicles (both cars and light duty trucks and vans). Among the "known" controls modeled by EPA are a number of behavioral or operational changes, such as enhanced inspection and maintenance requirements and carpooling programs. Beyond those relatively modest "known" costs come a substantial share of "unknown" control costs based primarily on early scrappage of the oldest personal vehicles. Households would also face costs for early scrappage of their non-road mobile equipment such as lawnmowers, snowmobiles, and ATVs.

- Manufacturing. As shown in Table D-1, a large fraction of the total costs for "known" controls represent measures for manufacturing sector sources, such as retrofitting boilers, steam generators, and other large stationary facilities with SCR, low-NO_X burners, and other emission-reduction equipment. In addition, cost inputs for this sector also reflect "unknown" controls that are likely a mix of retrofits on some of the smallest manufacturing sector point sources and retirements of some of the oldest combustion equipment rather than retrofitting it (the latter occurring to the extent that it has a lower cost-per-ton than the incremental retrofit on a relatively small source).

- Trucking. Costs to the trucking sector include retrofits of SCR onto heavy-duty trucks and operational changes such as anti-idling programs and electrification of truck stops. Nevertheless, a substantial share of the total costs is assumed to also involve early scrappage of the oldest, highest-emitting of the fleet of commercial trucks.

- Other. This category includes all other types of controls, each of which has a smaller total cost than any of the above categories. An important cost element in this column is retrofitting compressors along natural gas and oil pipelines with SCR (or possibly electrification at compressor locations where it may be more cost-effective), which we have estimated will cost pipeline companies about $22.5 billion (annualized). Controls for this sector also include replacing household space heaters, water heaters, and air conditions with lower-emitting (or non-emitting) versions, replacing agricultural equipment with lower-emitting (or non-emitting) versions, *etc.*

Costs to reduce electricity generation emissions by forcing coal-fired generating units to stop operating are also an important cost component in our analysis. These costs do not appear in Table D-3 because they were endogenously estimated by N_{ew}ERA, given a modeling constraint to eliminate generation from certain coal-fired units in certain states (*i.e.*, where those controls were a cost-effective part of a state's necessary NO_X tonnage reductions.) They account for approximately 8% of the total regulatory impact on GDP.

Table D-1. Cost Allocation for "Known" Controls to Source Categories and N_{ew}ERA Sectors: National Summary

EPA Source Category	Share of "Known" Control Costs	N_{ew}ERA Sector	Share of EPA Source Category Costs
EGU	-	Modeled	
Point (Non-EGU)	65%	Manufacturing	48%
		Refined Products	29%
		Services	17%
		Natural Gas	3%
		Crude Oil	3%
		Other	<1%
Area	12%	Services	75%
		Household Durable Goods	25%
Onroad Mobile	18%	Household Transportation	38%
		Trucking	31%
		Services	31%
Nonroad Mobile	5%	Commercial Trans (exc Trucking)	43%
		Manufacturing	25%
		Agriculture	13%
		Household Durable Goods	10%
		Services	9%

Note: State cost allocations reflect state-specific information and differ from national summary.
Source: NERA calculations as explained in text

Table D-2. Cost Allocation for "Unknown" Controls to Source Categories and N_{ew}ERA Sectors: National Summary

EPA Source Category	Share of "Unknown" Control Costs	N_{ew}ERA Sector	Share of EPA Source Category Costs
EGU	-	Modeled	
Point (Non-EGU)	10%	**Services**	56%
		Manufacturing	38%
		Crude Oil	2%
		Refined Products	2%
		Natural Gas	1%
		Coal	1%
		Other	<1%
Area	22%	Services	53%
		Natural Gas Pipelines	26%
		Household Durable Goods	18%
		Oil Pipelines	3%
Onroad Mobile	37%	Household Transportation	38%
		Trucking	31%
		Services	31%
Nonroad Mobile	30%	Commercial Trans (exc Trucking)	48%
		Manufacturing	24%
		Agriculture	12%
		Services	8%
		Household Durable Goods	8%

Note: State cost allocations reflect state-specific information and differ from national summary.
Source: NERA calculations as explained in text

Table D-3. Annualized Cost Inputs by State and N_{ew}ERA Sector for "Known" and "Unknown" Controls (millions of 2013$)

	Services	Commercial Trans (exc Trucking)	Household Transportation	Manufacturing	Trucking	Other	Total
U.S. Total	$109,994	$49,343	$48,275	$39,917	$39,776	$60,556	$347,862
Alabama	$29	-	-	$52	-	$8	$89
Arizona	$36	$5	$18	$7	$15	$5	$85
Arkansas	$12	-	$3	$14	$3	<$1	$32
California	$43,258	$19,403	$18,172	$16,165	$17,815	$14,336	$129,148
Colorado	$338	$115	$141	$140	$91	$622	$1,448
Connecticut	$2,334	$940	$1,137	$649	$372	$751	$6,184
Delaware	$94	$124	$67	$33	$38	$50	$406
Florida	$144	$61	$98	$104	$55	$52	$513
Georgia	$352	$109	$254	$246	$172	$73	$1,207
Idaho	$10	$3	$6	$6	$4	$3	$32
Illinois	$7,076	$2,810	$1,684	$2,367	$1,459	$3,754	$19,151
Indiana	$2,404	$910	$1,222	$2,419	$1,037	$1,635	$9,628
Iowa	-	-	-	-	-	-	-
Kansas	$7	-	-	<$1	-	<$1	$8
Kentucky	$35	$6	$14	$36	$14	$33	$139
Louisiana	$4,166	$9,921	$1,386	$794	$1,323	$7,673	$25,262
Maine	$36	$11	$10	$40	$8	$15	$120
Maryland	$3,841	$1,136	$2,091	$1,296	$1,479	$989	$10,833
Massachusetts	$524	$181	$141	$129	$88	$145	$1,209
Michigan	$2,664	$550	$1,482	$1,066	$922	$2,586	$9,269
Minnesota	-	-	-	-	-	-	-
Mississippi	-	-	-	-	-	-	-
Missouri	$435	$160	$125	$127	$286	$124	$1,258
Montana	-	-	-	-	-	-	-
Nebraska	$4	-	-	$4	-	<$1	$8
Nevada	$13	<$1	$3	<$1	$3	$3	$23
New Hampshire	-	-	-	-	-	-	-
New Jersey	$3,296	$1,238	$1,081	$907	$611	$909	$8,042
New Mexico	$37	-	<$1	$13	$1	$36	$88
New York	$14,435	$3,391	$4,028	$3,000	$3,654	$4,481	$32,989
North Carolina	$3,772	$779	$3,713	$2,156	$1,492	$1,191	$13,102
North Dakota	-	-	-	-	-	-	-
Ohio	$6,618	$2,276	$4,054	$2,598	$3,012	$2,269	$20,827
Oklahoma	$17	<$1	-	$39	-	$57	$113
Oregon	-	-	-	-	-	-	-
Pennsylvania	$7,794	$2,376	$4,371	$2,487	$2,779	$9,353	$29,160
Rhode Island	$79	$15	$25	$16	$12	$28	$174
South Carolina	$920	$340	$456	$469	$440	$259	$2,885
South Dakota	$28	$10	$10	$11	$7	$45	$112
Tennessee	$72	$12	$21	$56	$20	$20	$202
Texas	$3,350	$1,741	$1,577	$1,401	$1,846	$8,127	$18,043
Utah	$14	$1	$8	$6	$6	$8	$42
Vermont	-	-	-	-	-	-	-
Virginia	$377	$209	$212	$231	$167	$221	$1,417
Washington	$285	$259	$245	$131	$162	$127	$1,209
West Virginia	$69	-	$2	$13	$2	$2	$88
Wisconsin	$1,015	$248	$417	$692	$381	$552	$3,305
Wyoming	$4	-	<$1	<$1	<$1	$11	$16

Note: Table includes annualized costs of "known" and "unknown" controls except for compliance costs associated with control measures in the electric power sector (scrappage of coal-fired power plants), which are modeled in N_{ew}ERA. Estimates are based on annualized costs of "known" controls provided in EPA's 2008-2010 analyses and NERA state marginal cost curves (developed in annualized dollars per ton). See Figure D-3 and surrounding text for additional assumptions regarding annualization of compliance costs.

Source: NERA calculations as explained in text

The following sections describe the development of state-level sector shares for costs in each emission source category.

1. Point (Non-EGU)

Point (non-EGU) control costs were allocated to N_{ew}ERA sectors using three-digit North American Industry Classification System (NAICS) codes. For "known" controls we used costs reported by the EPA in their 2008-2010 ozone docket files for each control. We estimated the NAICS composition of "unknown" point control costs using EPA's 2018 projected state-level emissions remaining in each NAICS industry after applying any "known" point controls.

In most cases, the NAICS codes corresponded to a single N_{ew}ERA sector. Several codes, however, were divided among multiple N_{ew}ERA sectors to better reflect the composition of economic activity in the N_{ew}ERA macroeconomic model. This mapping of NAICS codes into N_{ew}ERA sectors is summarized in Table D-4. A few "known" controls did not include industry codes in the EPA data;[41] we placed costs for these controls in the services sector.

Controls in NAICS industry 211, Oil and Gas Extraction, were divided between the N_{ew}ERA natural gas and crude oil sectors using state-level shares of 2014 crude oil and natural gas production; for example, if a state produced natural gas but no crude oil in 2014, all of that state's Oil and Gas Extraction control costs were allocated to the natural gas sector in N_{ew}ERA. All other relationships between NAICS industry codes and N_{ew}ERA sectors were constant across states.

1. Non-Point (Area)

Costs for EPA "known" area source controls were split between the services sector (75%) and household durable goods (25%) in every state, as shown above in Table D-1. These splits roughly reflect the shares of "known" commercial/industrial area source controls and residential water and space heater controls shown in EPA's 2008 RIA (pp. 3a-4 to 3a-11).

Costs for "unknown" area controls were first divided between crude oil and natural gas pipeline controls and other area controls. In each state, the share of EPA projected baseline area source emissions attributable to oil and gas pipelines was used to approximate the share of state area source costs attributable to pipeline controls. Pipeline control costs were then split between the natural gas (90%) and crude oil sectors (10%), which affect delivered natural gas and oil prices in the N_{ew}ERA model. The remaining non-pipeline "unknown" area control costs were split between the services sector (75%) and household durable goods (25%).

[41] These represented less than 1% of annualized costs from "known" point source controls.

Table D-4. NAICS Codes Corresponding to N_{ew}ERA Sectors

NewERA Sector	NewERA Code	NAICS (1997)	NAICS Description	Share of NAICS
Agriculture	AGR	11 (111-115)	Agriculture, Forestry, Fishing and Hunting	
Natural Gas	GAS	211	Oil and Gas Extraction	Varies by state
Crude Oil	CRU	211	Oil and Gas Extraction	Varies by state
Coal	COL	212	Mining (except Oil and Gas)	30%
Refined Products	OIL	324	Petroleum and Coal Products Mfg	
Energy-Intensive Sectors	EIS	322	Paper Mfg	
		325	Chemical Mfg	
		327	Nonmetallic Mineral Product Mfg	80%
		331	Primary Metal Mfg	
Motor Vehicles	M_V	336	Transportation Equipment Mfg	50%
Other Manufacturing	MAN	212	Mining (except Oil and Gas)	70%
		213	Support Activities for Mining	
		23 (233-235)	Construction	
		327	Nonmetallic Mineral Product Mfg	20%
		336	Transportation Equipment Mfg	50%
		Other 31-33	Other Manufacturing	
Trucking	TRK	484-485	Truck, Transit, and Ground Passenger Trans	
Other Commercial Trans	TRN	481-483	Air, Rail, and Water Transportation	
Other Commerce & Services	SRV	22 (221)	Utilities	
		486	Pipeline Transportation	
		Other >33		

Source: NERA calculations as explained in text

2. Onroad

Onroad control costs for both "known" and "unknown" controls were allocated to N_{ew}ERA sectors using projected baseline onroad NO_X emissions in each state. The share of baseline onroad emissions attributable to passenger cars and light-duty trucks was used to approximate the share of state onroad control costs affecting household transportation in N_{ew}ERA. The remaining onroad control costs in each state were divided evenly between the trucking sector and the services sector.

3. Nonroad

Nonroad control costs for both "known" and "unknown" controls were similarly allocated to sectors using projected baseline nonroad NO_X emissions in each state. The share of baseline nonroad emissions attributable to marine and railroad sources was used to approximate the share of state nonroad control costs in the non-trucking commercial transportation sector. Remaining nonroad control costs were split between N_{ew}ERA sectors using industry shares of baseline projected emissions related to off-road equipment; these industries corresponded to non-trucking

commercial transportation, agriculture, services, manufacturing, and durable goods consumption in the N_{ew}ERA model.

4. Electricity Generating Units (EGUs)

EGU controls applied in EPA's 2008-2010 ozone analyses (based on IPM modeling) were not included in our compliance cost modeling. Instead, as discussed in Appendix C, we estimated the NO_X emission reductions and approximate electricity sector costs associated with coal power plant scrappage in each state within the N_{ew}ERA model. Whenever these coal scrappage controls are adopted by states to comply with a 60 ppb standard, the final sector-specific costs were determined endogenously in N_{ew}ERA.

B. Allocation of Costs to N_{ew}ERA Modeling Years

As discussed in the report body, the version of N_{ew}ERA used in this analysis models every third year between 2014 and 2038. This section describes the timing of compliance costs as N_{ew}ERA inputs based on estimated deadlines for ozone NAAQS attainment as well as the division of annualized costs into capital costs (incurred before compliance deadlines) and operating and maintenance (O&M) costs (incurred from the compliance deadlines onward).

1. Classifications and Attainment Timing

The ozone NAAQS have a staggered implementation. Areas in non-attainment of a standard are given classifications ranging from Marginal to Extreme based on recent ozone monitoring data. Each classification is associated with an attainment year such that areas with more severe classifications (higher ozone design values) are given additional time to come into attainment.

As discussed in Appendix B, several states would have nonattainment areas for the 1997 ozone standard of 84 ppb and the 2008 (current) standard of 75 ppb under future baseline conditions. We used existing EPA information to set the compliance deadlines for these two standards for our modeling. For 60 ppb compliance deadlines, we developed illustrative estimates of potential ozone design values for relevant states (those requiring reductions for 60 ppb according to the analysis in Appendix B) for 2016, the likely most recent data that will be available to EPA when it would be making its nonattainment designations for 60 ppb in 2017, as discussed below.

a. 1997 Standard of 84 ppb

Classifications and attainment years for the 1997 ozone standard of 84 ppb are still relevant for California and Texas; areas in these states were projected to be in nonattainment of 84 ppb in 2020 based on EPA's 2008-2010 analyses (and areas in these states have been above 84 ppb in ozone readings from recent years, as discussed in Section I of the report body). EPA assumed that California would have more time than other states to come into attainment with a new standard and estimated California reduction requirements and costs in 2030 (EPA 2008, p. 4-3).

We assumed 84 ppb attainment in 2030 for California and 2019 for Texas based on the attainment year for the Houston area (EPA 2013a).

b. 2008 Standard of 75 ppb

We developed state-level classifications for the current ozone standard of 75 ppb using EPA county-level classifications in EPA (2012 and 2013b). We use the most extreme classification of any county in a state to determine the classification and attainment year for the entire state. Attainment years for the different classifications are summarized in Table D-5.

Table D-5. Classifications and Attainment Years for the Current 75 ppb Standard

Classification	Attainment Year	States	
Compliant	N/A	22	AL, FL, ID, IA, KS, ME, MI, MN, MT, NE, NV, NH, NM, ND, OK, OR, RI, SD, UT, VT, WA, WV
Marginal	2016	23	AZ, AR, CO, CT, DE, GA, IL, IN, KY, LA, MA, MS, MO, NJ, NY, NC, OH, PA, SC, TN, VA, WI, WY
Moderate	2019	2	MD, TX
Serious	2022	0	
Severe-15	2028	0	
Severe-17	2030	0	
Extreme	2033	1	CA

Note: Attainment years represent the first year after a state comes into attainment. For example, EPA's attainment date for Marginal areas is December 31, 2015 (EPA 2012) and the NERA attainment year for Marginal states is 2016.

Source: EPA (2012) and NERA calculations as explained in text

c. Potential Standard of 60 ppb

For states requiring emission reductions for attainment with a 60 ppb standard based on our analysis of baseline and compliance emissions, we developed potential classifications using EPA's methodology for previous NAAQS and recent information on ozone concentrations and NO_X emissions.[42]

Our classifications are based on projected 2016 ozone design values as a percentage of the 60 ppb standard. Final classifications for a potential standard of 60 ppb would probably be determined by EPA in 2017. We assume that these classifications would be based upon three-

[42] Recent ozone monitoring data suggests that we may be understating the geographic scope of nonattainment under a potential 60 ppb standard. Several states projected to comply with a 60 ppb standard in our emissions analysis would be classified as nonattainment using recent ozone monitoring data and our classification methodology.

year averages of 4th-highest 8-hour ozone concentration monitor readings, so the relevant years of ozone concentrations would be 2014 through 2016.

We estimated these future ozone concentration averages (and percentage exceedances) by projecting county-level ozone readings in each state. We estimated background ozone by state ranging from 40 (in the eastern U.S.) to 55 (in the southwestern U.S.).[43] We then calculated NO_X emissions in future years by assuming a fixed growth rate from 2011 historical emissions to EPA's 2018 projected emissions, and we found for each state the percentage change in NO_X emissions from 2013 to the average for the 2014 through 2016 period. Finally, we applied this percentage change in the NO_X emissions to 2013 ozone monitor readings after netting out our estimate of the background ozone in the state.[44] These resulting adjusted ozone readings for 2014 through 2016 were the primary basis for our ozone classification of states.

EPA used percentage exceedance of the ozone standard to determine classifications for the 1997 and 2008 ozone standards (EPA 2012). We applied the same percentage exceedance cutoffs to classify states requiring emission reductions for 60 ppb. The high end of the ozone range for a classification of Marginal is 15% above 60 ppb, Moderate is 33.33% above, Serious is 50% above, Severe-15 is 58.33% above, Severe-17 is 133.33% above, and Extreme is everything higher. In the resulting classifications, 34 states are Marginal or Moderate, and Texas and Michigan are classified as Serious. California was "bumped up" to Extreme so that its classification for 60 ppb would not be lower than its classification for 75 ppb. Classifications for 60 ppb are shown in Table D-6.

We based the attainment year for each 60 ppb classification on expectations about rule timing and EPA's 2012 announcement regarding attainment timing for the current 75 ppb standard. The potential attainment years for 60 ppb classifications used in our analysis are summarized in Figure D-2.

We used the state classifications and attainment years for each standard as the basis for the assumed staggered implementation schedule in this analysis.

[43] These estimates are based on EPA (2014), Figure 7, p. 2A-28.

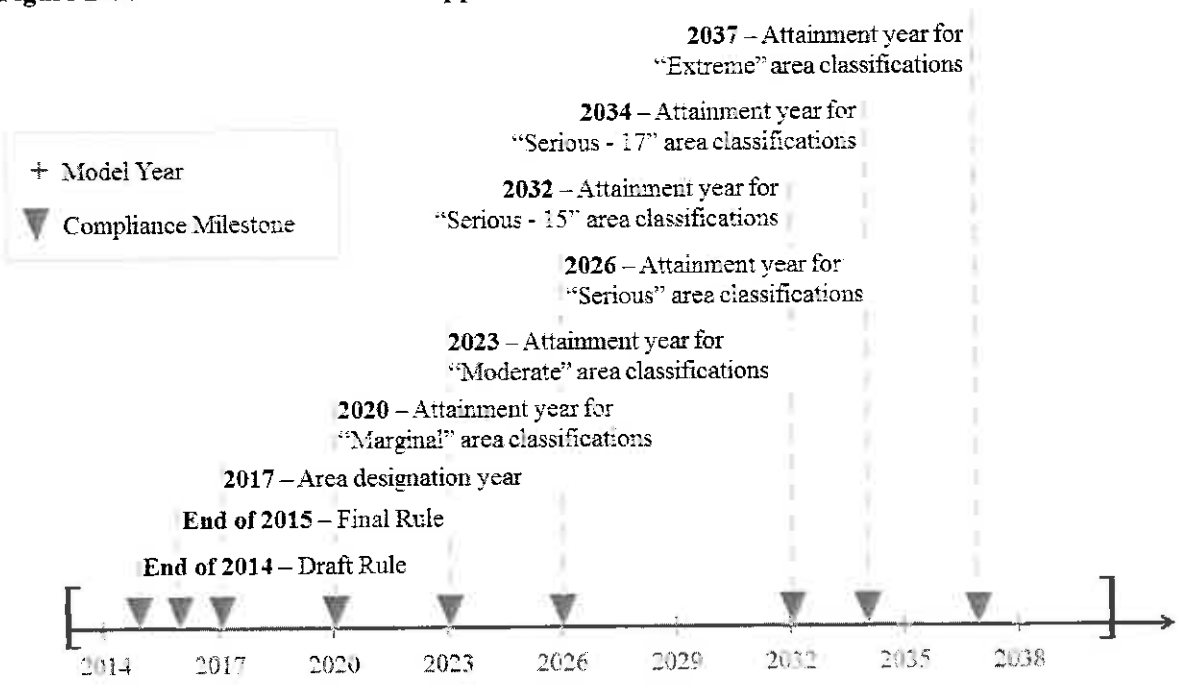
[44] For example, if the 2013 ozone reading was 70 ppb, the background was estimated to be 50 ppb, and the change in NO_X emissions between 2011 and 2018 was -20%, then we applied the -20% change to an ozone reading of 20 (70 less 50), which would be -4, and then added that back to the original ozone reading of 70 to arrive at an estimated 2014 through 2016 ozone reading of 66 ppb.

Table D-6. Classifications and Attainment Years for a Potential Standard of 60 ppb

Classification	Attainment Year	States	
Compliant	N/A	8	IA, MN, MS, MT, NH, ND, OR, VT
Marginal	2020	5	FL, NE, SC, SD, WA
Moderate	2023	32	AL, AZ, AR, CO, CT, DE, GA, ID, IL, IN, KS, KY, LA, ME, MD, MA, MO, NV, NJ, NM, NY, NC, OH, OK, PA, RI, TN, UT, VA, WV, WI, WY
Serious	2026	2	MI, TX
Severe-15	2032	0	
Severe-17	2034	0	
Extreme	2037	1	CA

Note: WA was classified as in attainment using our classification methodology but required controls based on our emissions analysis, so it was "bumped up" to a Marginal classification for cost timing purposes. CA was "bumped up" to an Extreme classification because it was Extreme for 75 ppb.
Source: NERA calculations as explained in text.

Figure D-2. Attainment Years for 60 ppb Classifications



Source: NERA calculations as explained in text
Note: The attainment years displayed in the figure represent the first year after a state comes into attainment.

2. Nature of Expenditures

a. Capital and O&M

The compliance cost estimates based on our state NO_X control marginal cost curves are annualized costs. We assumed 50% of annualized compliance costs are capital and 50% are operating and maintenance (O&M). The actual split will differ with the type of control; but controls generally require an up-front investment such as equipment retrofit/replacement and annual expenses to operate equipment or staff an ongoing program. We estimated the present value (*i.e.*, lump sum) of the capital portion of annualized compliance costs using 7% discount rate and 20-year capitalization period. The discount rate reflects EPA practice (EPA 2008, pp. 5-5 and 5-6), and the capitalization period reflects part of the range of equipment life for air emission controls used in EPA analyses (Pechan 2006). The capitalization period is an uncertain parameter in this analysis, and we would encourage EPA to provide detail on capitalization periods in its forthcoming ozone RIA.

b. Incremental Costs

Compliance costs were separated into incremental costs to achieve different ozone standards (84 ppb from baseline, 75 ppb incremental to 84 ppb, and 60 ppb incremental to 75 ppb). Costs attributable to each standard were then distributed over time according to state classifications and attainment years for the standards as described below.

c. VOC Costs

As discussed in Appendix C, EPA's 2008-2010 ozone analyses included the use of some VOC controls. These "known" VOC controls represent less than 1% of total compliance costs and we assume (following EPA) that NO_X is the limiting factor in ozone formation (EPA 2010, p. S2-3); however the ozone impact of NO_X reductions in EPA's analysis was incremental to these VOC controls, so we include them in our total compliance costs when reasonable for consistency. If we estimate that a state incurs any NO_X control costs to comply with the existing standard of 75 ppb, we include any EPA-modeled VOC control costs for that state using the same timing assumptions as other compliance costs attributable to the 75 ppb standard. If we estimate that a state is already compliant with a 75 ppb standard but requires reductions and costs to attain a 60 ppb standard, we apply VOC control costs in that state according the timing assumptions for the 60 ppb standard. Timing assumptions for the different ozone standards are described below.

3. Cost Timing

a. Distribution of Capital Costs

The present value of capital costs for each state and standard was distributed in an increasing fashion over the years leading up to attainment under the assumption that capital costs would be incurred before the compliance deadline for each state. We assumed capital costs would be

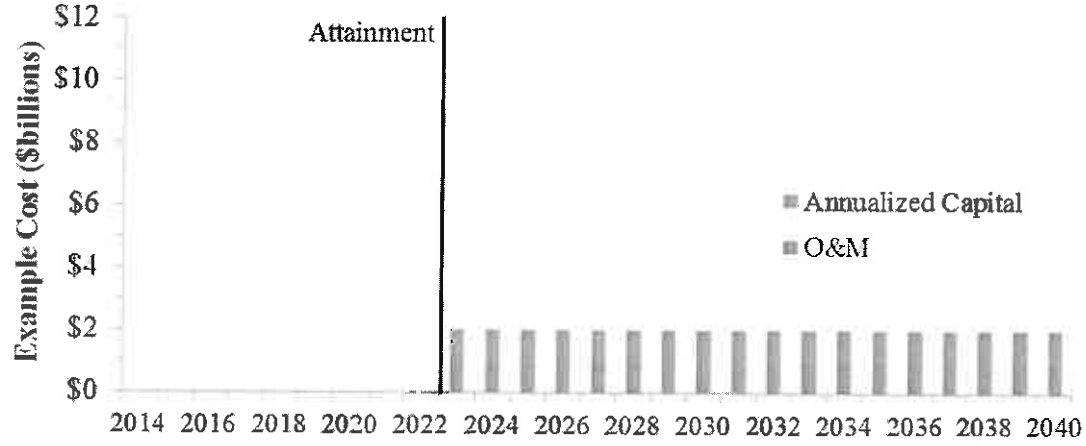
incurred up to nine years prior to attainment.[45] Costs attributable to the potential standard of 60 ppb were incurred no earlier than 2017 (allowing time for EPA to develop state designations).

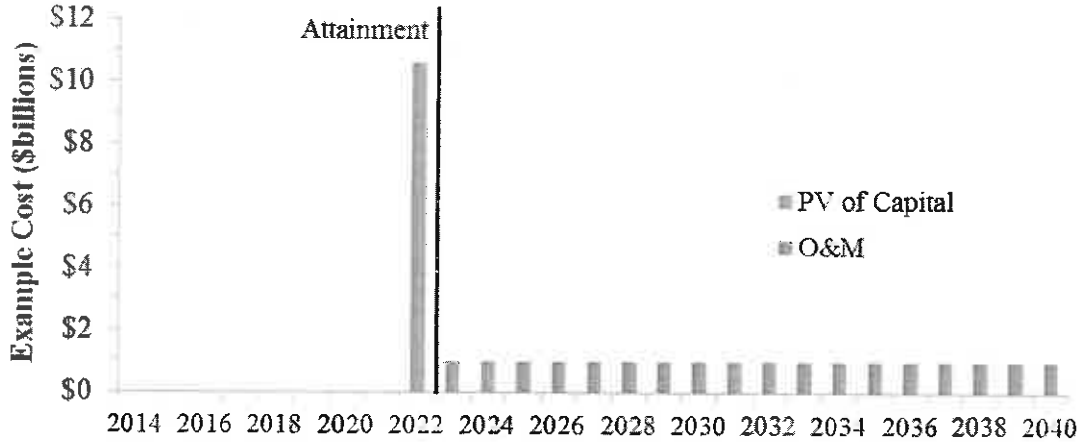
We divided capital costs so that they would be small shares of the total present value in early years and large shares of the total present value in the years immediately prior to attainment of a standard. Specifically, we counted the number of years over which capital would be distributed for a certain state and standard; for example, capital costs for Moderate states to comply with a 60 ppb standard (incremental to 75 ppb) were distributed over six years from 2017 through 2022 (the year prior to attainment). We then calculated the sum of the numbers up to the compliance year (*e.g.*, 1 + 2 + 3 + 4 + 5 + 6 = 21), and assigned shares of costs to each year based on its number relative to the sum (1/21 = 5% for 2017, 2/21 = 10% for 2018, *etc.*).

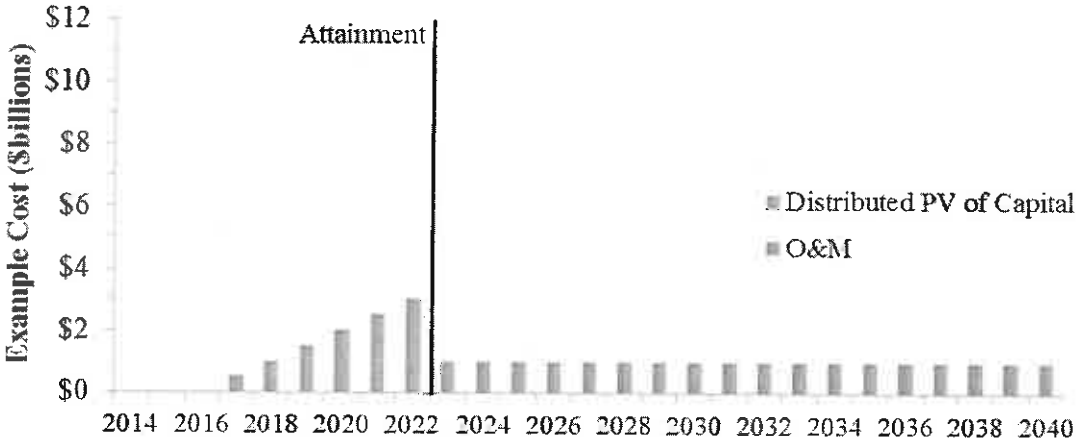
Figure D-3 provides an example of our capital cost distribution methodology using an area classified as Moderate for 60 ppb with $1 billion of annualized capital costs beginning in the attainment year (2023). In this example, we calculate the present value of the 20-year stream of annualized capital costs one year before attainment (in this case 2022) using a 7% discount rate. This present value ($10.6 billion) is shown in the second graph. Finally, we spread the present value over the years leading up to attainment, following the method discussed above.

[45] In one case, the 84 ppb standard in California, we spread capital costs over more than nine years (the entire period from 2014 to the assumed attainment year of 2030).

Figure D-3. Illustration of Capital Cost Distribution Methodology





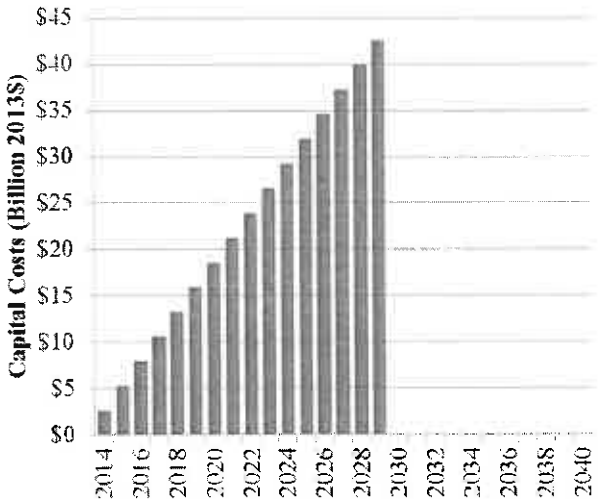


Note: Example based on a state classified as Moderate for 60 ppb, implying an attainment year of 2023 and no capital costs incurred prior to 2017.
Source: Illustrative example

Note that capital costs for different standards may overlap depending on specific state classifications and attainment years. For example, California was designated as Extreme under the 84 ppb standard with an attainment year of 2030; we also designate California as Extreme under a 75 ppb standard with an attainment year of 2033 and Extreme under a 60 ppb standard with an attainment year of 2037. California will incur capital costs for 84 ppb from 2014 through 2030, it will incur capital costs for 75 ppb from 2024 through 2032 (the nine years prior to attainment), and it will incur capital costs for 60 ppb from 2028 through 2036 (again, the nine years prior to attainment).

The distributions of nationwide capital costs for each incremental ozone standard are summarized in Figure D-4 through Figure D-6.

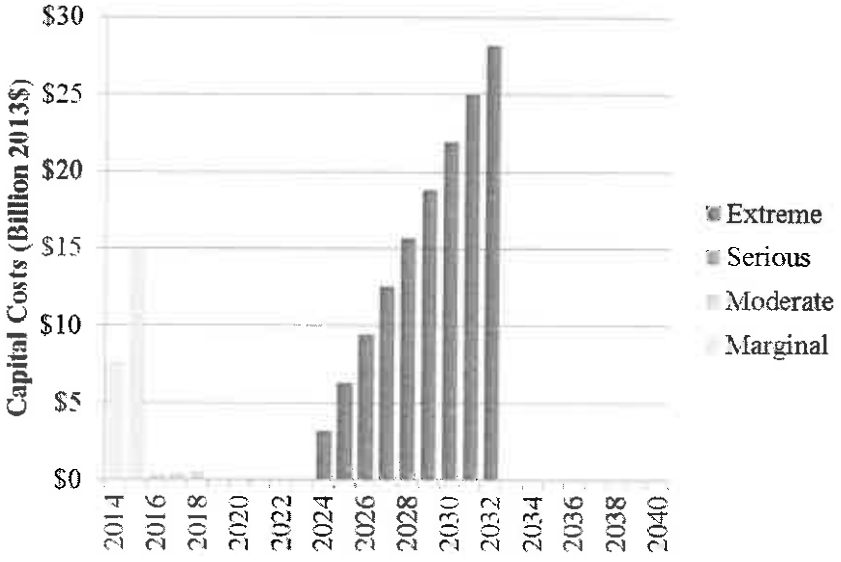
Figure D-4. Capital Costs by Year to Attain 84 ppb



Note: All costs to attain 84 ppb in our analysis are in California. Texas also requires reductions to attain 84 ppb, but due to difficulty assembling detailed EPA information on "known" controls in Texas we conservatively assume zero costs for Texas to attain 84 ppb (as noted in Appendix C).

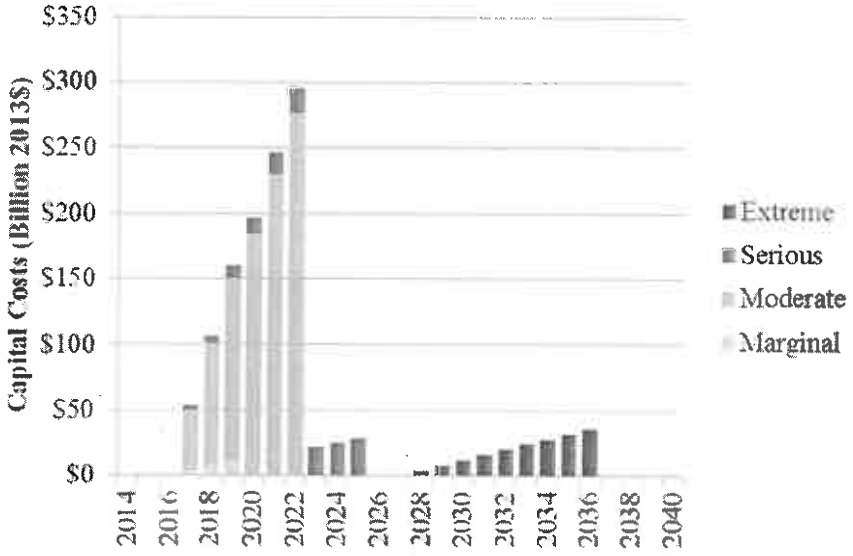
Source: NERA calculations as explained in text.

Figure D-5. Nationwide Capital Costs by Year to Attain 75 ppb (Incremental to 84 ppb)



Source: NERA calculations as explained in text.

Figure D-6. Nationwide Capital Costs by Year to Attain 60 ppb (Incremental to 75 ppb)



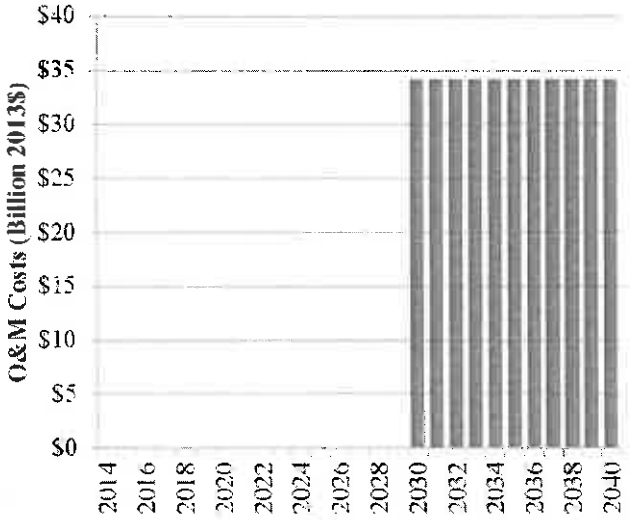
Source: NERA calculations as explained in text

b. Distribution of O&M Costs

We assume that the O&M portion of total annualized compliance costs under each standard are incurred in every year from the relevant state attainment year to the end of the model period.

The distributions of nationwide O&M costs for each incremental ozone standard are summarized in Figure D-7 through Figure D-9.

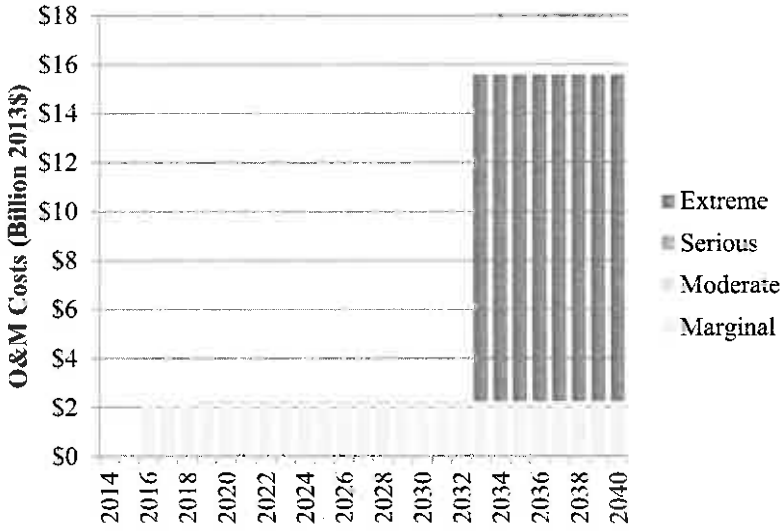
Figure D-7. O&M Costs by Year to Attain 84 ppb



Note: All costs to attain 84 ppb in our analysis are in California. Texas also requires reductions to attain 84 ppb, but due to difficulty assembling detailed EPA information on "known" controls in Texas we conservatively assume zero costs for Texas to attain 84 ppb (as noted in Appendix C).

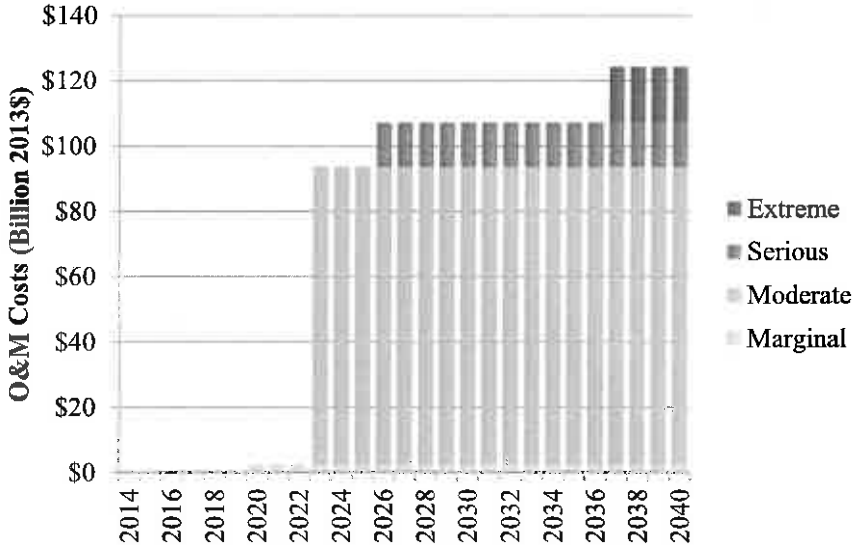
Source: NERA calculations as explained in text.

Figure D-8. Nationwide O&M Costs by Year to Attain 75 ppb (Incremental to 84 ppb)



Source: NERA calculations as explained in text.

Figure D-9. Nationwide O&M Costs by Year to Attain 60 ppb (Incremental to 75 ppb)



Source: NERA calculations as explained in text.

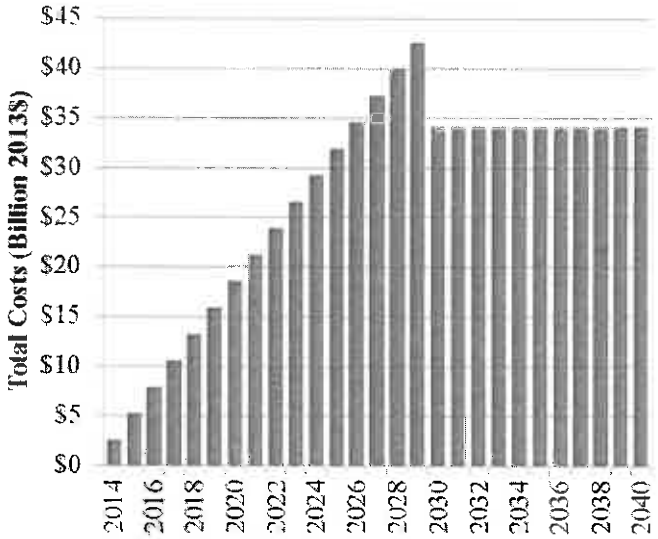
c. Total Costs by Year

The distributions of *nationwide* total costs (capital and O&M) for each incremental ozone standard are summarized in Figure D-10 through Figure D-13.

d. Total Costs by N_{ew}ERA Model Year

The N_{ew}ERA model used for this analysis operates in three-year increments beginning in 2014; model year 2014 represents 2014 through2016, model year 2017 represents 2017 through2019, and so on. For each model year, we took averages of annual compliance cost estimates for the corresponding three-year increment to create N_{ew}ERA input costs. Figure D-14 provides a national summary of those state-level N_{ew}ERA input costs.

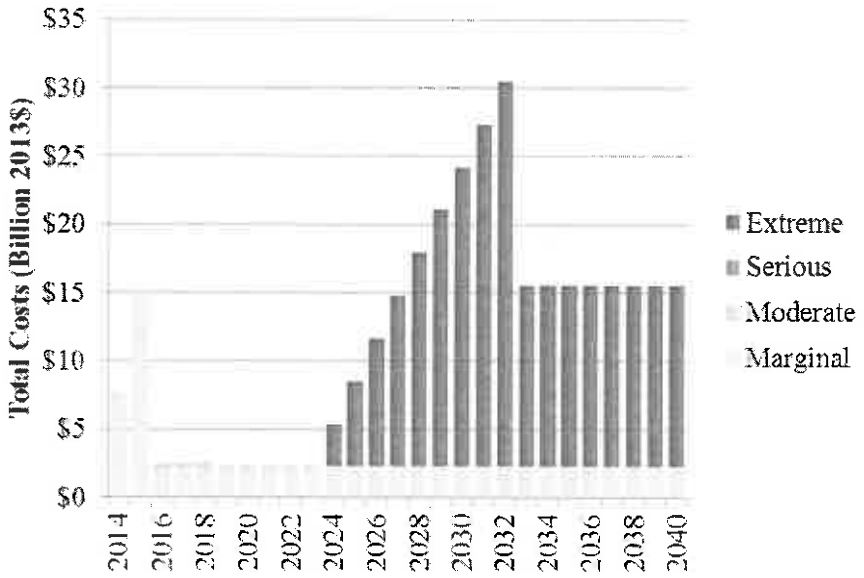
Figure D-10. Total Costs by Year to Attain 84 ppb



Note: All costs to attain 84 ppb in our analysis are in California. Texas also requires reductions to attain 84 ppb, but due to difficulty assembling detailed EPA information on "known" controls in Texas we conservatively assume zero costs for Texas to attain 84 ppb (as noted in Appendix C).

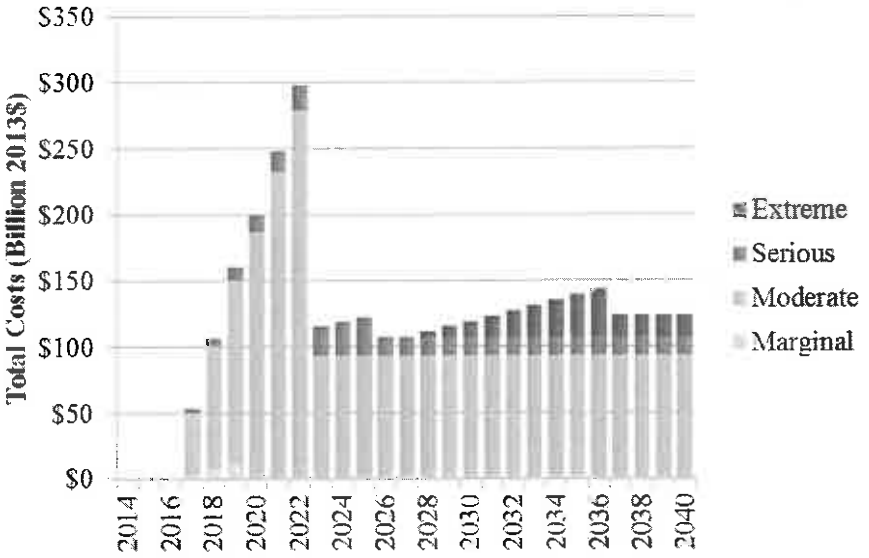
Source: NERA calculations as explained in text.

Figure D-11. Nationwide Total Costs by Year to Attain 75 ppb (Incremental to 84 ppb)



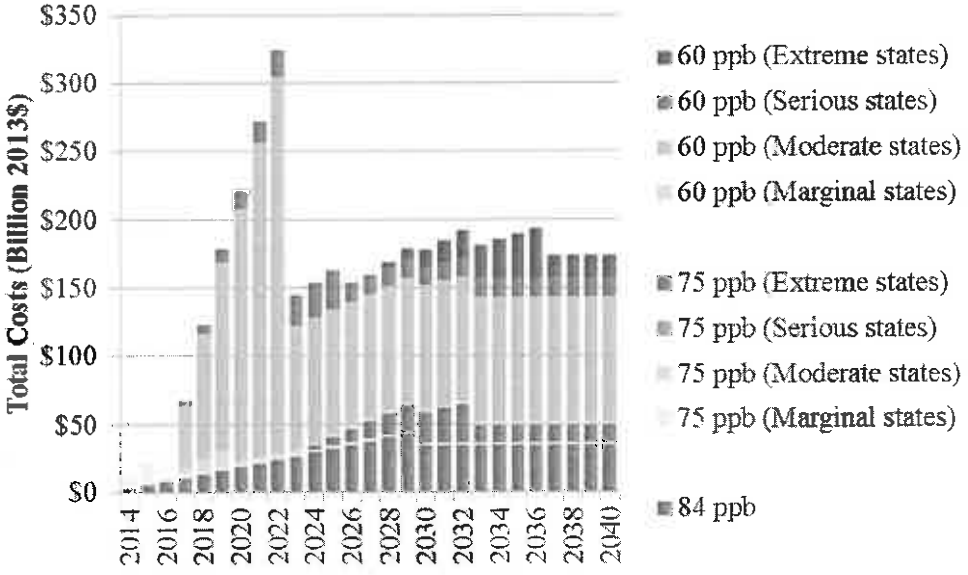
Source: NERA calculations as explained in text.

Figure D-12. Nationwide Total Costs by Year to Attain 60 ppb (Incremental to 75 ppb)



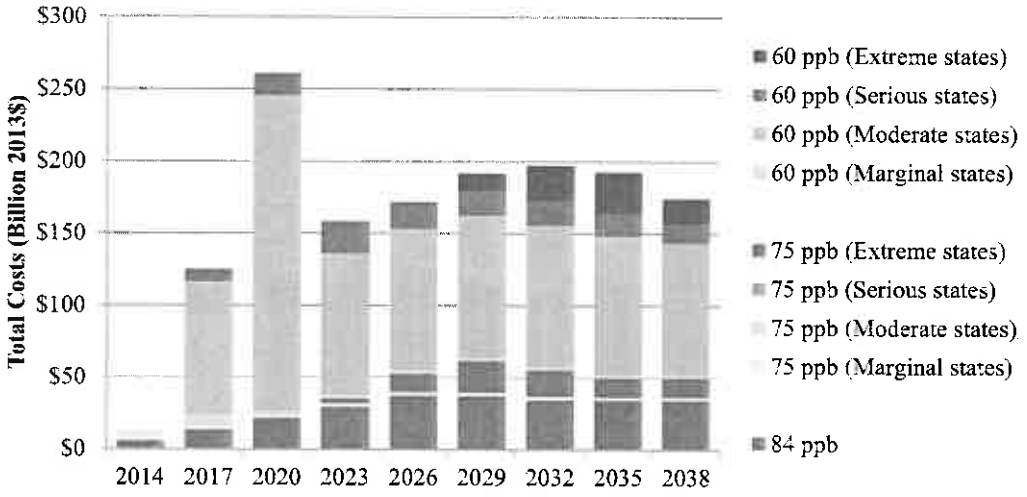
Source: NERA calculations as explained in text.

Figure D-13. Nationwide Total Costs by Year and Standard



Source: NERA calculations as explained in text.

Figure D-14. Nationwide Total Costs by N_{ew}ERA Model Year and Standard



Source: NERA calculations as explained in text.

C. References to the Appendix

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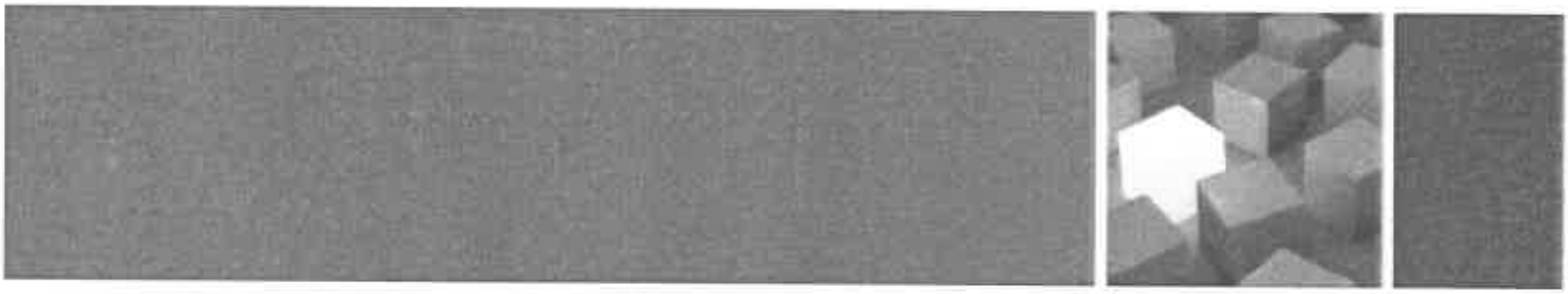

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Appendix H
Economic Impacts of a 65 ppb NAAQS for Ozone

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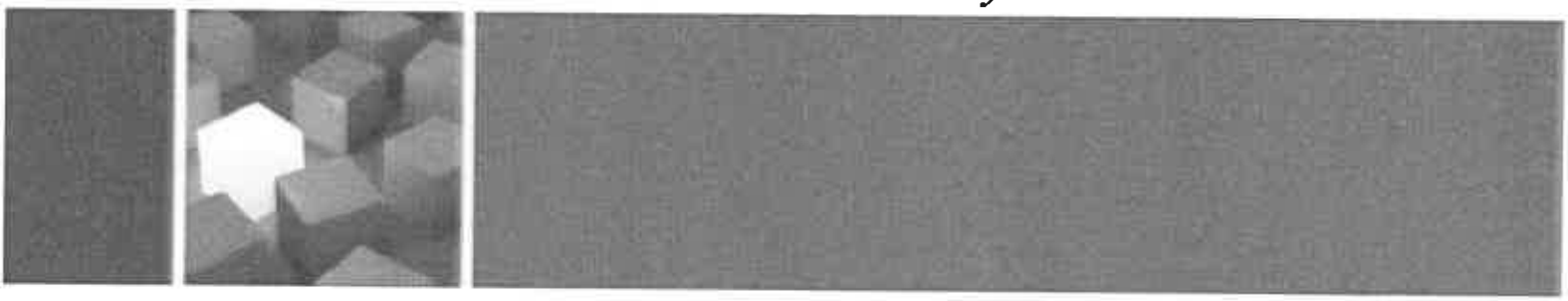
Economic Impacts of a 65 ppb National Ambient Air Quality Standard for Ozone

Executive Summary



Prepared for:

National Association of Manufacturers

February 2015

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EXECUTIVE SUMMARY

This study evaluates the potential compliance costs and impacts on the U.S. economy if the U.S. Environmental Protection Agency (EPA) were to set a National Ambient Air Quality Standard (NAAQS) for ozone of 65 parts per billion (ppb). Employing our integrated energy-economic macroeconomic model (N_{ew}ERA), we estimate that the potential emissions control costs could reduce U.S. Gross Domestic Product (GDP) by about $140 billion per year on average over the period from 2017 through 2040 and by about $1.7 trillion over that period in present value terms.[1] The potential labor market impacts represent an average annual loss employment income equivalent to 1.4 million jobs (*i.e.*, job-equivalents).[2]

These results represent updated values from the results in our July 2014 report (NERA 2014), which developed estimates of the potential costs and economic impacts of achieving a 60 ppb ozone standard using the best information then available. In November 2014, the U.S. Environmental Protection Agency (EPA) released updated emissions and cost information supporting their proposal to revise the ozone standard (EPA 2014a); we have used that new information to update our analysis. Also, given that the proposed rule suggests setting a revised ozone NAAQS in the range of 65 ppb to 70 ppb, in this update we assess the economic impacts of a potential 65 ppb ozone NAAQS. This Executive Summary of our study begins with a summary of the differences between the information and methodology in our July 2014 report and those used in this updated study. It then provides summaries of our estimates of the costs and economic impacts of attaining a potential ozone NAAQS of 65 ppb.

Changes in Data and Methodology Since the July 2014 Report

The methodology used for this study is largely similar to the methodology used in our July 2014 report. This section discusses changes to the three components of our analysis:

1. *The methodology for estimating emission reductions.* This study used updated EPA information on the future NO_X and VOC emissions levels needed to comply with a potential 65 ppb standard (rather than a 60 ppb standard as in our July 2014 report).

---

[1] All dollar values in this report are in 2014 dollars unless otherwise noted. The present value reflects impacts from 2017 through 2040, as of 2014 discounted at a 5% real discount rate; this discount rate falls in the 3% to 7% range recommended in EPA's *Guidelines for Preparing Economic Analyses* (2010a, p. 6-19), and it is consistent with the discount rate used in the N_{ew}ERA model.

[2] "Job-equivalents" is defined as total labor income change divided by the average annual income per job. This measure does not represent a projection of numbers of workers that may need to change jobs and/or be unemployed, as some or all of the loss could be spread across workers who remain employed, thereby impacting many more that 1.4 million workers, but with lesser impacts per worker.

Additionally, we used updated cost and effectiveness information about emission controls that have been identified by EPA.

2. *The methodology for estimating compliance costs.* We updated the costs of the known controls that EPA identified to attain the 65 ppb standard using EPA's new cost data. However, even for a 65 ppb standard, more than half of the emissions reductions needed across the country would come from measures that EPA still has not identified. Using the same evidence-based approach for developing a cost curve that we used in our July analysis (but using the more recent inventory data, and updating the calculations for a later year of compliance spending), we calculated the costs of the set of further emissions reduction needs that EPA has left unidentified in its current analysis. We also updated all dollar figures from 2013 to 2014 dollars.

3. *The methodology for estimating economic impacts.* We used the same version of NERA's N_{ew}ERA macroeconomic model as our previous study to estimate the economic impacts of our estimated costs for reducing emissions in the amount necessary to attain a 65 ppb ozone standard. In contrast to EPA's analysis, we excluded the proposed EPA Clean Power Plan rule from our modeling baseline.

In our July 2014 report, we performed a sensitivity analysis on the possibility that nonattainment, especially in rural areas of the U.S., could create barriers to continued growth in oil and gas extraction. A national policy question that remains in a state of flux is whether or not new permitting requirements hinder growth in energy production. A tightened ozone standard has the potential to cause nonattainment areas to expand into relatively rural areas, where there are few or no existing emissions sources that could be controlled to offset increased emissions from new activity. If nonattainment expands into rural areas that are active in U.S. oil and gas extraction, a shortage of potential offsets may translate into a significant barrier to obtaining permits for the new wells and pipelines needed to expand (or even maintain) our domestic oil and gas production levels. The sensitivity analysis in our July 2014 report resulted in much larger natural gas price effects, and raised macroeconomic impacts of our base case by about 30 to 50%. Limitations of time have prevented us from conducting a similar sensitivity analysis for this update.

Methodology for Estimating Emission Reductions

The July 2014 report relied on projected 2018 baseline VOC and NO_X emissions and EPA information from its 2008 and 2010 Regulatory Impact Analyses (RIAs) to estimate reductions required for all regions of the U.S. to come into compliance with a 60 ppb standard. The updated EPA information that we rely on in this study includes projected 2018 and 2025 base case and baseline emissions as well as EPA's estimates of reductions required from the 2025 baseline emissions to achieve a 65 ppb standard (EPA 2014a-g). We use the updated EPA estimates of

state-by-state emissions reductions from the 2025 baseline as the principal basis for our estimates of NO_X emissions levels that would allow a 65 ppb standard to be attained nationwide.[3] In order to reach and maintain this level of NO_X emissions consistent with a 65 ppb ozone concentration, states would need to reduce emissions at existing sources and prevent any *net* increases in emissions from new or expanded sources. We also rely on EPA's revised data on the cost of emissions reductions for "known" control measures, which are provided by source sector and state.

Our methodology for estimating costs of emission reductions is similar to our July 2014 study. In both studies, we substituted our base case estimates of electricity generating unit (EGU) emissions for those of EPA, for consistency with our economic impact model, which estimates costs from EGU emissions reductions endogenously. As before, we adopted EPA's cost estimates for those controls that EPA identifies as "known"—that is specific controls for which EPA had developed emission reduction and cost information—and we applied our own more evidence-based approach for estimating costs for the many required reductions that EPA treats as "unknown." For estimating the impacts to the U.S. economy of our estimates of compliance costs, we assigned each state's projected cost to specific calendar years, using assessments of their likely attainment dates. Also consistent with our prior study, we assigned the costs to specific sectors in each state; for the "known" control measures these assignments were based on the sector-specific information available in EPA's data and for the "unknown" control measures, these assignments were based on emissions inventory data on the relative contribution of each source category to the remaining emissions in each state.

Methodology for Estimating Compliance Costs

Our methodology for developing estimates of compliance costs in this study is the same as in our July 2014 report, although of course the numerical values are different reflecting the additional information now available. As noted, EPA developed updated estimates of the annualized costs from "known" controls, and we used this updated information on "known" controls.

As in the July 2014 analysis, emission reductions from "known" controls were not sufficient to achieve attainment, in this case with a 65 ppb ozone standard. EPA has filled the gap with a rough estimate of costs of "unknown" controls, *i.e.*, controls for which no cost information was developed. In contrast to the two cost estimation methodologies presented in its 2008 and 2010 RIAs, this time EPA used a single simplistic assumption that annualized control costs for these "unknown" controls would be equal to $15,000 per ton, regardless of the state, the sector, or the amount of emission reduction required. This estimate was not based upon any evidence-based

[3] We focused our analysis on NO_X emissions, but we also included EPA's estimates of VOC emission control costs in our modeling.

analyses of the nature of the emissions that remain after "known" controls are in place, or of the costs of potential additional controls for these sources.

Our compliance cost estimates are based upon a synthesis of EPA estimates of emission reduction, our modifications of EPA's assumptions regarding baseline reductions, EPA's estimates of the costs of "known" controls, and our more detailed estimates of the costs of "unknown" controls. As in our July 2014 report, our "unknown" cost estimates are more evidence-based than EPA's, as we use detailed information on the types of sources that account for the remaining emissions (EGUs, other point sources, on-road sources, off-road mobile sources, and area sources) as well as estimates of the potential costs of reducing emissions by scrapping existing emission sources prematurely. We updated our estimates of the costs of scrapping light-duty motor vehicles using up-to-date information. We also used updated information to assess the implications of these dollar-per-ton values for the marginal cost curve for reductions needed to achieve compliance. As in the July 2014 study, the result is a set of estimates of the costs for each state to comply with a more stringent ozone standard based upon the use of specific information to assess "unknown" control costs.

Methodology for Estimating Economic Impacts

Our methodology for estimating economic impacts of the estimated costs of compliance with a 65 ppb ozone standard is the same as in the July 2014 study for a 60 ppb standard, using NERA's N_{ew}ERA macroeconomic model. In the N_{ew}ERA model, expenditures on emissions control measures to comply with a new ozone standard reduce investment in other productive sectors of the economy, which results in decreases in economic output in subsequent years. The capital costs associated with compliance spending are assumed to be incurred from 2017 until 2036 (the last projected compliance date, for extreme areas), while each state's estimated operating and maintenance (O&M) costs are incurred for all years after the state's attainment date. Our economic impact analysis accounts for the effects of costs projected to be incurred through 2040.

N_{ew}ERA is an economy-wide integrated energy and economic model that includes a bottom-up, unit-specific representation of the electric sector, as well as a representation of all other sectors of the economy and households. It assesses, on an integrated basis, the effects of major policies on individual sectors as well as the overall economy. It has substantial detail for all of the energy sources used by the economy, with separate sectors for coal production, crude oil extraction, electricity generation, refined petroleum products, and natural gas production. The model performs its analysis with regional detail. As discussed above, this particular analysis uses state-specific cost inputs, and N_{ew}ERA has been run to assess economic impacts for each state. Appendix A of the July 2014 report provides a detailed description of the N_{ew}ERA model.

The macroeconomic analysis requires a baseline that projects economic outcomes in the absence of the incremental spending to attain the tighter ozone NAAQS. For this study, N_{ew}ERA's

baseline conditions were calibrated to reflect projections developed by Federal government agencies, notably the Energy Information Administration (EIA) as defined in its *Annual Energy Outlook 2014* (*AEO 2014*) Reference case. This baseline includes the effects of environmental regulations that have already been promulgated as well as other factors that lead to changes over time in the U.S. economy and the various sectors. Our baseline does not include the effects of proposed regulations, such as the Clean Power Plan (CPP), although we do include power sector closures as an available way to attain the NAAQS, to the extent that we find such closures to be cost-effective elements of each state's control strategy.[4]

The July 2014 report and appendices provide details on the various aspects of our methodology, subject to the changes noted above. Although this Executive Summary report describes results for the United States as a whole and disaggregated to 11 regions,[5] the inputs and the results are built up using detailed state-specific and sector-specific cost information. The costs and impacts of a more stringent ozone standard differ substantially among states.

Summary of National Results

Emission Reductions Required to Achieve a 65 ppb Ozone Standard

As Figure S-1 illustrates, national NO_X emissions have already been reduced substantially, from about 25.2 million tons in 1990 to 12.9 million tons in 2013 (EPA 2014b). EPA currently projects that U.S. NO_X emissions will be further reduced by existing rules and regulations to 8.2 million tons by 2025 (supplemented with N_{ew}ERA's projected baseline EGU emissions, which does not include the proposed CPP). Those additional emissions reductions between 2013 and 2025 will involve costs beyond the compliance costs estimated in this study. Economic activity (as measured by real GDP) in 2025 is projected to be more than double the level in 1990 (CEA 2014, Table B-3 and OMB 2013, Table 2), suggesting that U.S. NO_X sources will have been controlled by more than 80% by 2025, without the additional controls needed to attain a tighter ozone NAAQS.

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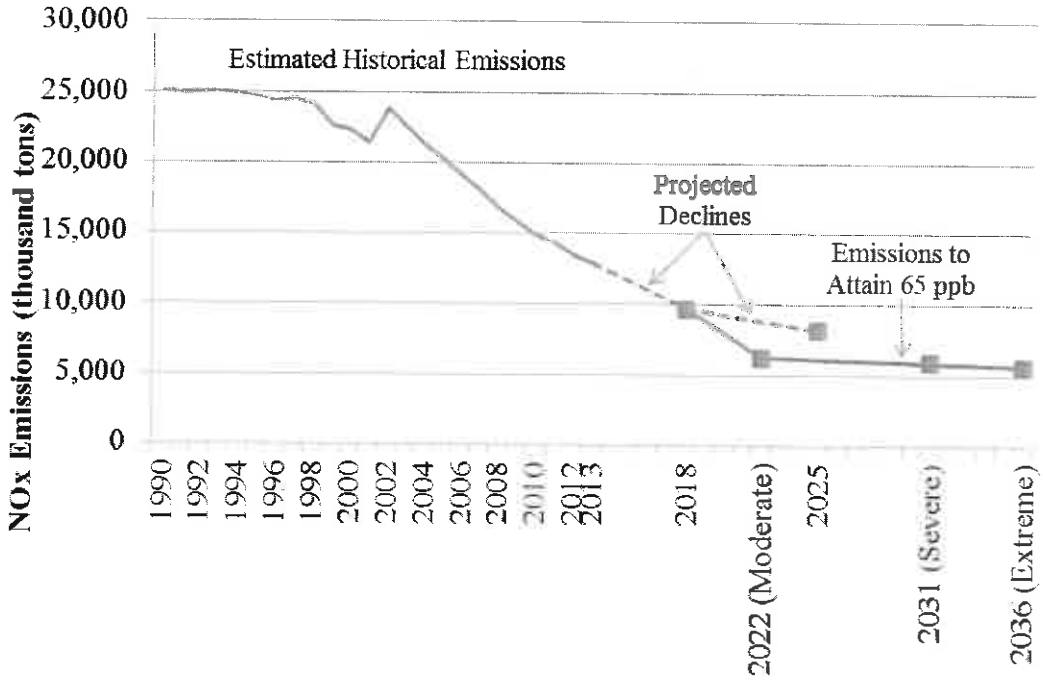
[4] EPA's inclusion of the CPP in its baseline was inconsistent with its standard practice of only including promulgated regulations. This deviation from standard procedure seems particularly unjustified given the enormous uncertainty in what carbon limits may actually be applied and how states would comply, and hence what NO_X emission reductions might actually occur as a result of this carbon regulation.

[5] "U.S." results are, formally, only for the lower 48 states, and exclude Alaska and Hawaii, as well as Washington DC. We refer to the lower 48 states as "U.S." hereafter.

Figure S-1: U.S. NO_X Emissions to Attain 65 ppb NAAQS Compared to Historical NO_X



Notes: Blue solid line: Estimated historical emissions.
Blue dotted line: Projected further declines through 2018 and 2025 (linear interpolation).
Red line: Emissions to attain 65 ppb on attainment schedule, with states not requiring reductions for 65 ppb held constant after 2025.
The slight increase in U.S. NO_X emissions from 2001 to 2002 primarily reflects changes in EPA's emission modeling methodology for onroad and nonroad sources (switching from MOBILE6 to the National Mobile Inventory Model and MOVES)

Source: NERA calculations as explained in text

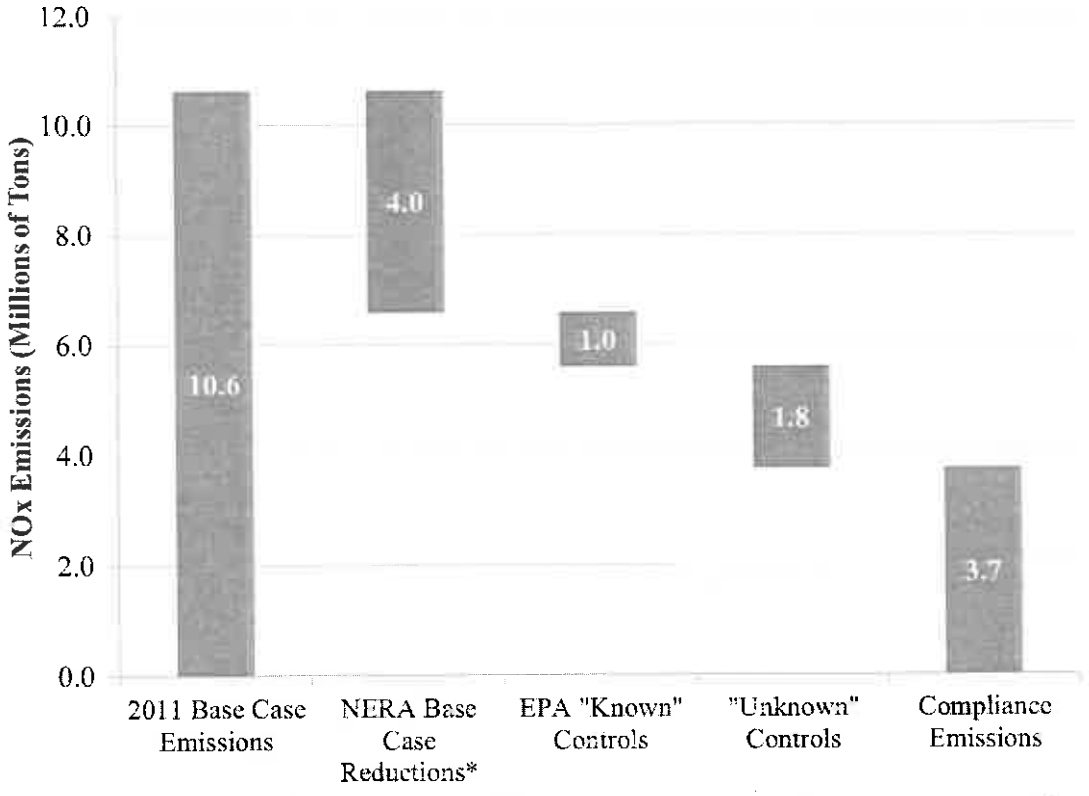
Based on the EPA information, total U.S. NO_X emissions would have to be reduced to about 6.2 million tons by 2022 and 5.6 million tons by 2036 to meet a 65 ppb standard throughout the nation. This reduction appears as the red line above in Figure S-1, which also shows our prognosis of the timing of those reductions, based on our estimates of the likely severity classifications of the different states.[6]

Figure S-2 shows our estimates of emissions and emission reductions for the 34 states that would not attain a 65 ppb under baseline conditions. Despite the extensive controls already expected to

[6] Nonattainment areas are given different classifications—marginal, moderate, serious, severe or extreme—depending on how far out of attainment they are with the NAAQS at the time that designations must be made, two years after promulgation.

occur in the future, we estimate that about 2.6 million additional tons (in aggregate) would need to be eliminated by 2022 and an additional 300,000 tons would need to be eliminated by 2036 in order for those states to come into attainment on schedule. This is equivalent to roughly another 25% reduction from the reduction estimated solely based on those states' 2025 NO_X emissions. It implies almost a 90% total reduction from all sizes and types of NO_X-emitting sources from the relatively uncontrolled emissions rates in 1990 (after adjusting for growth).

Figure S-2: NO_X Emissions and Categories of NO_X Reductions to Attain 65 ppb NAAQS (for 34 Non-Attaining States Only)



Note: Emissions and reductions include only states requiring emission reductions for compliance with a new ozone NAAQS of 65 ppb in this analysis.
*The NERA Base Case reflects 2022 conditions in each state requiring reductions, with two exceptions: The Base Case for UT and CA reflect conditions in 2031 and 2036, respectively, based on higher likely severity classifications in those two states.

Source: NERA calculations as explained in text

Figure S-3 shows the mix of emission reductions needed across 34 states that EPA projects will face compliance costs to achieve a 65 ppb ozone standard, including our estimates of the allocation of "unknown controls" to individual source categories. The dark green shows EPA's

"known controls" and the light green shows NERA's evidence-based assumptions regarding where "unknown controls" will likely come from.[7] The remaining sum (shown in the blue bars) is 3.7 million tons—the aggregate limit for those 34 states to achieve attainment in all the states projected to be in nonattainment under baseline conditions. This 3.7 million ton aggregate limit needs to be met by the attainment deadlines, which we assume to be 2022 for all states except California and Utah, which are assumed to have much later attainment dates.[8]

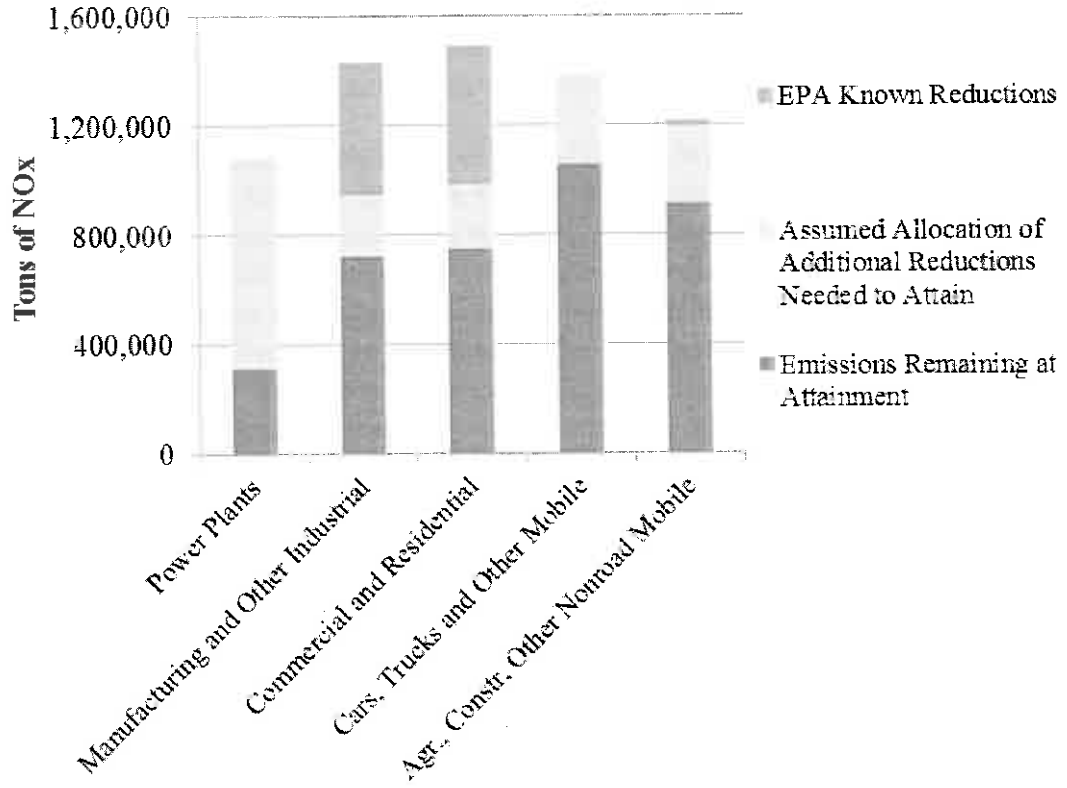
As noted above, NERA's estimates of what the "unknown" controls will comprise includes deep cuts in the EGU sector, where emissions are concentrated in a few sources and costs per ton are thus lower than for the many smaller sources among the non-point source categories (*i.e.*, area, onroad mobile and nonroad mobile). NERA estimates that the remaining "unknown" controls outside of the EGU sector will involve much smaller incremental percentage reductions than from EGUs, because these will require programs such as scrapping a portion of vehicles and other small sources. These controls are also projected to come at a substantially higher cost per ton than the EGU controls—even though we assume that the small-source scrapping programs will only target the oldest, highest-emitting of each type of NO_X-emitting equipment.[9]

[7] This figure does not show the amount of EGU controls (mostly from installation of SCRs) that EPA has identified as "known" control in that sector because our analysis shows that one of the most cost-effective forms of control that EPA has called "unknown" will be to close those EGUs instead. Thus, we assume that the SCRs in EPA's list of "known" controls will not actually be installed, and replace their reductions with the much larger reductions that would come from EGU closures that are cost-effective for meeting a 65 ppb NAAQS (which appear as the light green area on the EGU bar).

[8] States that will be classified as marginal nonattainment in 2017 will face a 2020 attainment date, or will be re-designated as moderate, and then must be in attainment by 2023. Our analysis suggests that some of the marginal states may reach attainment by 2020 without incremental controls other than the baseline reductions, and they face no compliance cost in our analysis. We have assumed that marginal states that would not attain by 2020 under their baseline forecast will not undertake early costly action to avoid reclassification as moderate, and will attain by the moderate attainment date along with states that will have been classified as moderate in 2017.

[9] For example, our estimates of costs and tons removed by scrappage of light-duty cars is limited to vehicles still on the road in 2022 that are of a pre-2008 model year (*i.e.*, pre-Tier 2 vehicles). We estimate that those older vintages of cars will account for about 40% of projected light-duty vehicle emissions in 2022.

Figure S-3: NERA Analysis's Allocation of Additional Reductions Necessary to Attain a 65 ppb NAAQS to Categories of Emissions Sources in the 34 Non-Attaining States



Source: NERA calculations as explained in text

Compliance Costs to Achieve a 65 ppb Ozone Standard

We estimate that the potential costs of achieving a 65 ppb ozone standard could have a present value of almost $1.1 trillion as of 2014 (based upon costs incurred from 2017 through 2040), not including any costs for forcing a massive cutback in generation from coal-fired EGUs to reduce NO_X emissions from the power sector (whose costs are endogenously determined in the economic impact model).[10] These costs are reported in Figure S-4. As a rough point of

[10] Although the precise costs of the EGU closures is determined in the model, we used preliminary model runs to identify which closures would be as or more cost-effective than other unknown controls in our analysis. Based on this exercise, we estimate that the majority of the NO_X emission reductions associated with the EGU closures cost an average of about $16,000 per ton, and range well above $30,000 per ton in some states. The result of the constraints that we applied was 34 GW of outright unit retirements, but a substantial number of additional GW of coal-fired capacity is left on-line but no longer generates in the model. This means that more than 34 GW is effectively closed down in our analysis.

comparison, we estimate that EPA's annualized cost estimate implies a present value of about $167 billion.[11] The primary difference in our methodologies is the extrapolation method used to estimate the cost of "unknown" controls; we attempted to assess the kinds of controls that would be required after "known" controls and based our method on the estimated costs per ton of one such control (vehicle scrappage), whereas EPA relied on an arbitrary constant value.

Figure S-4: Potential U.S. Compliance Spending Costs for 65 ppb Ozone Standard

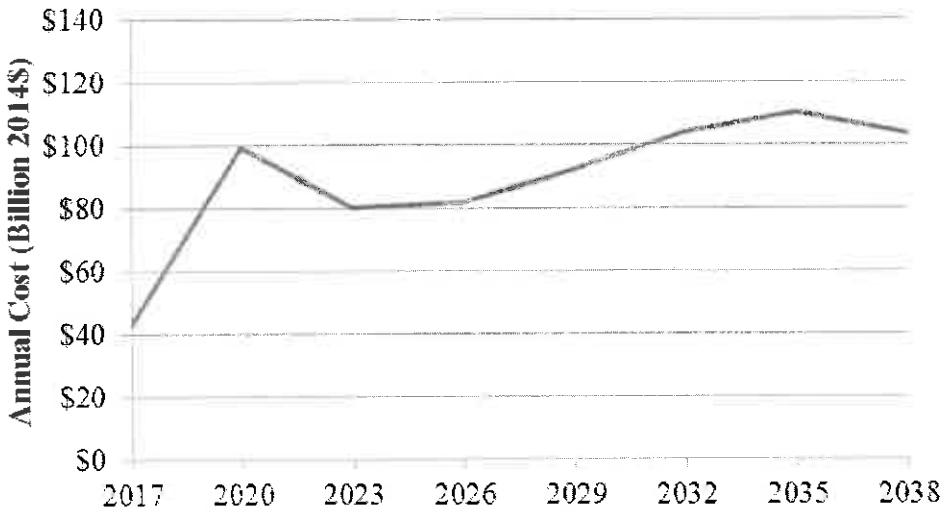
	Present Value (Billions of 2014$)			Cumulative
	Capital	O&M	Total	Coal Retirements
Compliance Costs	$430	$630	$1,050	34 GW

Notes: Total is not equal to the sum of capital and O&M due to independent rounding. Present value is from 2017 through 2040, discounted to 2014 at a 5% real discount rate.
Cumulative coal retirements are incremental to baseline. These retirements are primarily due to assumed emission control measures but may also include indirect electric sector impacts of the ozone standards. This number is understated because it reflects only those plants that the model literally closes, while substantial additional GW of coal unit capacity is not reported by the model as "retired" but nevertheless is forced into a position of near-zero utilization.
Source: NERA calculations as explained in text

Allocating the estimated capital costs to spending in years prior to each state's projected compliance deadline, and allocating O&M costs to years after the respective compliance deadlines, Figure S-5 shows the pattern of annual compliance spending across all states (except for the endogenously-determined costs of coal unit retirements.)

[11] This estimate assumes that EPA's total annualized cost estimate of $17 billion (including California) is incurred over a period of 20 years; that these 20 years begin in 2020, except in California where they begin in 2030; that these annual costs are converted to a present value in 2014 using a real annual discount rate of 5%; and that the present value is converted from 2011 dollars to 2014 dollars. Note that there are many differences in the EPA and NERA calculations so this estimate can only be viewed as providing a rough comparison.

Figure S-5: Potential Annual U.S. Compliance Spending Costs for 65 ppb Ozone Standard



Notes: Figure does not include compliance costs associated control measures in the electric power sector (scrappage of coal-fired power plants), which are modeled in N_{ew}ERA.
Source: NERA calculations as explained in text

Potential Impacts on the U.S. Economy and U.S. Households

The potential costs we estimated for a 65 ppb ozone standard are projected to have substantial impacts on the U.S. economy and U.S. households. Figure S-6 shows the potential macroeconomic effects as measured by GDP and U.S. household consumption. The 65 ppb ozone standard is projected to reduce GDP from the baseline levels by about $1.7 trillion on a present value basis from 2017 to 2040 (as of 2014, and in 2014 dollars) and by $140 billion per year on a levelized average basis over that period (*i.e.*, when spread evenly over years but retaining the same present value). Average annual household consumption over those same years could be reduced by an average of about $830 per household per year.

Figure S-6: Potential Impacts of 65 ppb Ozone Standard on U.S. Gross Domestic Product and Household Consumption

	Annualized	Present Value
GDP Loss (Billions of 2014$)	$140/year	$1,720
Consumption Loss per Household (2014$)	$830/year	N/A

Notes: Present value is from 2017 through 2040, discounted at a 5% real discount rate. Consumption per household is an annualized (or levelized) value calculated using a 5% real discount rate.
Source: NERA calculations as explained in text

Figure S-7 focuses on several dimensions of projected impacts on income from labor ("worker income") as a result of the 65 ppb ozone standard. Relative to baseline levels, real wages decline by about 0.6% on average over the period and labor income declines by about 0.9% on average, resulting in job-equivalent losses that average about 1.4 million job-equivalents. (Job-equivalents are defined as the change in labor income divided by the annual baseline income for the average job (see Figure S-7)). A loss of one job-equivalent does not necessarily mean one less employed person—it may be manifested as a combination of fewer people working and less income per worker. However, this measure allows us to express employment-related impacts in terms of an equivalent number of employees earning the average prevailing wage.[12] These are the *net* effects on labor and include the positive benefits of increased labor demand in sectors providing pollution control equipment and technologies.

Figure S-7: Potential Impacts of 65 ppb Ozone Standard on Labor

	Avg.
Baseline Annual Job-Equivalents (millions)	156
65 ppb Case:	
Real Wage Rate (% Change from Baseline)	-0.6%
Change in Labor Income (% Change from Baseline)	-0.9%
Job-Equivalents (Change from Baseline, millions)	-1.4

Notes: Average (Avg.) is the simple average over 2017-2040. "Job-equivalents" is defined as total labor income change divided by the average annual income per job. This measure does not represent a projection of numbers of workers that may need to change jobs and/or be unemployed, as some or all of the loss could be spread across workers who remain employed

Source: NERA calculations as explained in text

Potential Effects on U.S. Energy Prices

Emissions reduction costs of a 65 ppb ozone standard also is likely to have impacts on U.S. energy sectors, largely because the more stringent ozone standard is projected to lead to the premature retirement of many additional coal-fired power plants. Figure S-8 shows average energy price projections under the baseline and the 65 ppb ozone standard. The average delivered residential electricity price is projected to increase by an average of 1.7% over the period from 2017 through 2040 relative to what they could otherwise be in each year (which is

[12] The N_{ew}ERA model, like many other similar economic models, does not develop projections of unemployment rates or layoffs associated with reductions in labor income. Modeling such largely transitional phenomena requires a different type of modeling methodology; our methodology considers only the long-run, equilibrium impact levels.

projected to be rising even without a tighter ozone NAAQS). Henry Hub natural gas prices are projected to increase by an average of 3.7% in the same time period (again, relative to what they could otherwise be in each future year), while delivered residential natural gas prices could increase by an average of 3.7%. Part of the increase in delivered natural gas prices reflects the increase in pipeline costs due to control costs for reductions in NO_X emissions in the pipeline system that could be recovered through tariff rates.

Figure S-8: Potential Impacts of a 65 ppb Ozone Standard on Energy Prices Relative to Their Projected Levels in Each Future Year

		Avg. Baseline	Avg. 65 ppb Case	Change	% Change
Henry Hub Natural Gas	$/MMBtu	$6.22	$6.47	$0.25	3.7%
Natural Gas Delivered (Residential)	$/MMBtu	$14.23	$14.76	$0.53	3.7%
Natural Gas Delivered (Industrial)	$/MMBtu	$8.71	$9.27	$0.55	6.3%
Gasoline	$/gallon	$3.68	$3.69	$0.01	0.3%
Electricity (Residential)	¢/kWh	14.9¢	15.2¢	0.2¢	1.7%
Electricity (Industrial)	¢/kWh	9.7¢	10.0¢	0.3¢	2.8%

Notes: Average is the simple average over 2017-2040. The Baseline reflects expected growth in prices over the analysis period as predicted by the *Annual Energy Outlook 2014*. Figures in 2014$.

Source: NERA calculations as explained in text

Potential Effects on U.S. Sectors and Regions

All sectors of the economy would be affected by a 65 ppb ozone standard, both directly through increased emissions control costs and indirectly through impacts on affected entities' customers and/or suppliers. There are noticeable differences across sectors, however. Figure S-9 and Figure S-10 show the estimated changes in output for the non-energy and energy sectors of the economy, respectively, due to the emissions reduction costs of a 65 ppb ozone standard.

Figure S-9: Potential Impacts of 65 ppb Ozone Standard on Output of Non-Energy Sectors (Percentage Changes from Baseline)

	Agriculture	Commercial/ Services	Manufacturing	Commercial Transportation	Commercial Trucking
Average (2017-2040)	-0.9%	-0.4%	-0.3%	-0.9%	-0.5%

Note: Values are the simple average of percentage change over 2017-2040.
Source: NERA calculations as explained in text

Figure S-10: Potential Impacts of a 65 ppb Ozone Standard on Output of Energy Sectors (Percentage Changes from Baseline)

	Coal	Natural Gas	Crude Oil/Refining	Electricity
Average (2017-2040)	-28%	3.9%	-0.8%	-1.5%

Note: Values are the simple average of percentage change over 2017-2040.
Source: NERA calculations as explained in text

Figure S-11 shows the estimated average annual change in consumption per household for individual N_{ew}ERA regions. A region's attainment costs and its sectoral output mix determine to a large extent whether a region fares better or worse than the U.S. average, but all regions could experience lower household consumption.

Figure S-11: Potential Impacts of a 65 ppb Ozone Standard on Annual Consumption per Household by Region

Region	2014$
Arizona and Mountain States	-$690
California	-$790
Florida	-$250
Mid-America	-$770
Mid-Atlantic	-$1,370
Mississippi Valley	-$640
New York/New England	-$1,530
Pacific Northwest	-$310
Southeast	-$620
Texas, Oklahoma, Louisiana	-$1,290
Upper Midwest	-$490
U.S.	**-$830**

Notes: Values are the levelized average over 2017-2040, annualized using a 5% real discount rate.
Maps of NewERA regions are provided in the report body and Appendix A.

Source: NERA calculations as explained in text

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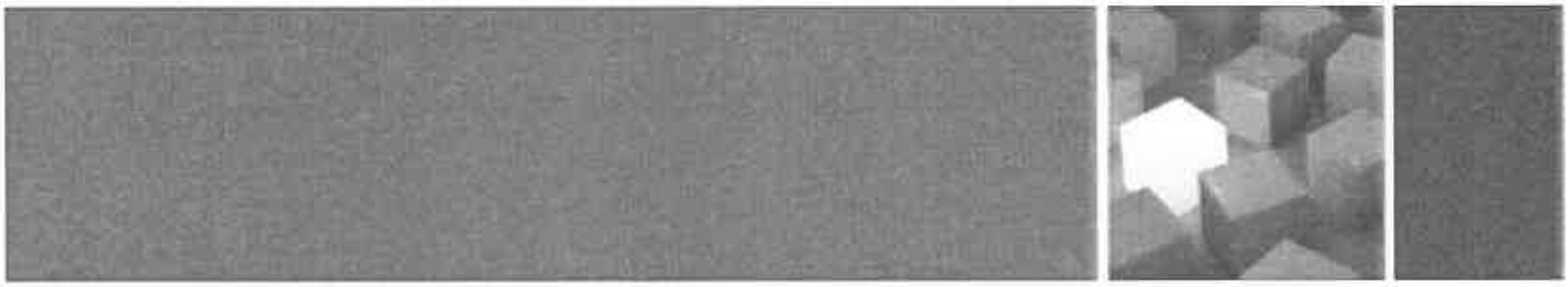

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Appendix I
What Could New Ozone Regulations Cost Ohio?

What Could New Ozone Regulations Cost Ohio?

$22 Billion Gross State Product Loss from 2017 to 2040

22,914 Lost Jobs or Job Equivalents[1] per Year

$840 Million in Total Compliance Costs

$440 Drop in Average Household Consumption per Year

[1] Total job equivalents equal total labor income change divided by the average annual income per job.

Expensive New Ozone Regulation Will Put the Squeeze on Ohio

The Environmental Protection Agency's (EPA) new ozone regulation could be the most expensive ever issued on the American public, costing the nation $140 billion annually,[1] according to a new analysis by NERA Economic Consulting. This regulation will make it harder to get the necessary permits to manufacture goods and build critical infrastructure like roads and highways in Ohio, while increasing the cost of energy for every business and household in the state. The picture gets even worse for the counties in the red and orange in the maps in figure 1. In these areas, manufacturers won't be able to expand without a reduction of emissions or shutdown of operations from other plants in the area. Plans for new plants and expansion at existing plants will be shelved. Federal highway funds could freeze and economic growth could grind to a halt.

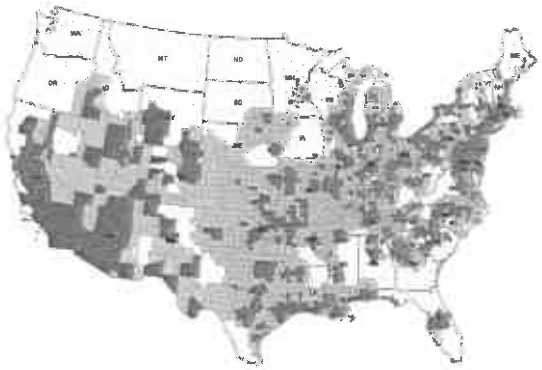
[1] This estimate only accounts for the costs and related economic impacts of bringing the country into attainment with a 65 ppb ozone standard. It does not account for any additional costs incurred by businesses complying with "maintenance" requirements for attainment areas. This estimate also does not account for any potential curtailment of energy production in nonattainment areas. In NERA's July 2014 report measuring a 60 ppb ozone standard, they found that a significant curtailment of natural gas production in nonattainment areas could further reduce GDP by $90 billion per year and cost an additional 1.4 million job-equivalents per year.

www.nam.org/ozone

Figure 1: Projected Nonattainment with a 65 Parts Per Billion (ppb) Ozone Standard

- Areas with monitors
- Unmonitored but likely to exceed 65 ppb

Projected Nonattainment in Ohio (65 ppb)



Projected Nonattainment in the United States (65 ppb)

Source: URS

National Association of Manufacturers

Ozone Levels:
Counties in Ohio

County	Ozone Levels (ppb)
Brown County	81
Butler County	81
Clermont County	81
Hamilton County	81
Warren County	81
Cuyahoga County	80
Delaware County	80
Fairfield County	80
Franklin County	80
Geauga County	80
Hocking County	80
Lake County	80
Licking County	80
Lorain County	80
Madison County	80
Medina County	80
Morrow County	80
Perry County	80
Pickaway County	80
Union County	80
Clinton County	78
Mahoning County	77
Trumbull County	77
Carroll County	76
Greene County	76
Highland County*	76
Miami County	76
Montgomery County	76
Stark County	76
Ashtabula County	75
Clark County	75

Table Key:

Highlighted Counties = Nonattainment at 65ppb

* = Based on Interpolation

Source: EPS, July 3, 2014, Based on 3-year period, 2011-2013

Ozone Levels:
Counties in Ohio

County	Ozone Levels (ppb)
Tuscarawas County*	75
Champaign County*	74
Adams County*	73
Allen County	73
Auglaize County*	73
Coshocton County*	73
Darke County*	73
Fulton County	73
Holmes County*	73
Knox County	73
Logan County*	73
Lucas County	73
Ottawa County*	73
Ross County*	73
Shelby County*	73
Williams County*	73
Wood County	73
Crawford County*	72
Fayette County	72
Hardin County*	72
Jefferson County	72
Marion County*	72
Mercer County*	72
Preble County	72
Putnam County*	72
Richland County*	72
Ashland County*	71
Columbiana County*	71
Erie County*	71
Hancock County*	71
Henry County*	71

Table Key:

Highlighted Counties = Nonattainment at 65ppb

* = Based on Interpolation

Source: URS, July 3, 2014. Based on 3-year period 2011-2013

Ozone Levels:
Counties in Ohio

County	Ozone Levels (ppb)
Muskingum County*	71
Sandusky County*	71
Seneca County*	71
Wayne County*	71
Wyandot County*	71
Belmont County	70
Defiance County*	70
Harrison County*	70
Huron County*	70
Pike County*	70
Van Wert County*	70
Lawrence County	69
Paulding County*	69
Vinton County*	69
Washington County	69
Athens County*	68
Gallia County*	68
Jackson County*	68
Meigs County*	68
Morgan County*	68
Portage County	68
Scioto County*	68
Summit County	68
Monroe County*	67
Guernsey County*	66
Noble County	66

Table Key:

Highlighted Counties = Nonattainment at 65ppb

* = Based on Interpolation

Source: URS, July 3, 2014. Based on 3-year period, 2011-2013