Memorandum

TO: Docket for rulemaking: "National Emission Standards for Hazardous Air Pollutants: Coal- and Oil-Fired Electric Utility Steam Generating Units – Subcategory of Certain Existing Electric Utility Steam Generating Units Firing Eastern Bituminous Coal Refuse for Emissions of Acid Gas Hazardous Air Pollutants" (EPA-HQ-OAR-2018-0794)

DATE: April 8, 2020

SUBJECT: Analysis of Potential Costs and Benefits for the "National Emission Standards for Hazardous Air Pollutants: Coal- and Oil-Fired Electric Utility Steam Generating Units – Subcategory of Certain Existing Electric Utility Steam Generating Units Firing Eastern Bituminous Coal Refuse for Emissions of Acid Gas Hazardous Air Pollutants"

1. <u>Introduction</u>

This memorandum, conducted pursuant to Executive Order 12866 and 13563, provides information related to the potential costs and benefits associated with establishing an emissions subcategory for Eastern Bituminous Coal Refuse (EBCR)-fired Electric Utility Steam Generating Units (EGUs).[1] The subcategorization establishes emissions rate limits for hydrochloric acid (HCl) and sulfur dioxide (SO_2), both of which serve as surrogates for emissions rates of all acid gas hazardous air pollutants (HAP).[2] As discussed in the preamble for this final rule, on February 7, 2019, the EPA solicited comment on subcategorizing the MATS acid gas emission standards for these units.[3] In this memo, we present quantitative estimates of the compliance cost reductions, forgone emissions reductions, and forgone benefits associated with the subcategorization under a primary baseline. The memo also presents a qualitative discussion of the potential impacts of the subcategorization for a second, alternative baseline. The memo provides a discussion of key uncertainties associated with our estimates of the costs and benefits of the subcategorization.

The primary baseline assumes that the affected units meet the acid gas HAP emissions rate requirement established in the 2012 Mercury and Air Toxics Standards 77 FR 9304 (2012 MATS). Consistent with the 2012 MATS regulatory impact analysis (RIA), we assume in the primary baseline that these units will install and operate dry scrubbers (*i.e.*, spray dryer absorbers or SDA) in order to comply with the 2012 MATS acid gas HAP emissions limits. While we assume that the baseline for this analysis is compliance with the acid gas limits in the 2012 MATS rule, the dry scrubbers we assume in the analysis to be needed to comply with the acid gas limits have not been installed at these units as of the signature of this final action. The policy case in this scenario assumes compliance with the emissions rate requirement established by the

[1] Hereafter referred to as "the subcategorization."
[2] Other acid gas HAP include hydrogen fluoride (HF), hydrogen cyanide (HCN), and selenium dioxide (SeO_2).
[3] See 84 FR at 2700-2703. EPA did not solicit comment on establishing subcategory emission standards for non-acid gas HAP (mercury, non-mercury metals, and organic HAP).

subcategorization with no other change in their operation over time (*i.e.*, emission controls that may be needed under the 2012 MATS limits need not be installed).

The uncertainty about what the affected units would do in the short term to comply if the EPA did not finalize the EBCR subcategory, in addition to the uncertainty about future economic conditions and utilization under any scenario, make a multiyear analysis highly speculative. As a result, the illustrative cost and benefit estimates for the primary scenario reflect analysis for a representative year of 2023.[4] The single year analysis presented in the memo provides a reasonable snapshot of possible impacts at a given point in time, 2023, the year for which we have access to air quality modeling information useful for this rule.

All monetized costs and benefits in this memo are denominated in 2016 dollars. Data and spreadsheets supporting the analysis in this memo are included in the docket.[5]

The second scenario presented in this memo qualitatively discusses potential environmental and economic impacts that could result under an alternative baseline in which the 2012 MATS acid gas HAP standards absent the subcategorization lead to unit closures rather than continued operation and investment in air pollution control equipment as assumed in the primary scenario. Certain affected EBCR-fired EGUs and others submitted comments indicating that some of these units may discontinue operation absent the subcategorization finalized in this action.[6] The policy case with this alternative scenario is the same as with the primary scenario. This second scenario focuses on qualitative impacts because the potential impacts of the units closing absent the subcategory are difficult to fully quantify. It is however an important scenario to consider because, unlike traditional coal-fired EGUs whose primary purpose is to generate electricity, coal refuse-fired EGUs serve two important purposes in that the units generate electricity and remediate coal refuse piles, which provides air quality and water quality benefits. The coal refuse piles can spontaneously self-ignite and combust (smolder) internally. Uncontrolled burning of coal refuse piles releases uncontrolled air emissions, unlike burning the refuse in a coal refuse-fired boiler. The use of coal refuse-fired power plants as a remediation technique has received policy support for many years in Pennsylvania, the state where most of these plants are located. This policy support, which includes both tax credits and alternative energy credits, indicates the value the state of Pennsylvania places on addressing this issue.

There are six affected EGUs in the subcategory, located at four power plants in Pennsylvania and West Virginia (Table 1).[7] The units have low generating capacities relative to other fossil-steam

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[4] As we assume units will install dry scrubbers absent the subcategory, we assume the control equipment will be operated over multiple years.

[5] The document containing the analysis presented in this memorandum is titled Data and Spreadsheets Supporting the Analysis of Potential Costs and Benefits for the "National Emission Standards for Hazardous Air Pollutants: Coal- and Oil-Fired Electric Utility Steam Generating Units – Subcategory of Certain Existing Electric Utility Steam Generating Units Firing Eastern Bituminous Coal Refuse for Emissions of Acid Gas Hazardous Air Pollutants".

[6] See, for example, Docket ID Numbers. HQ-OAR-2018-0794-1154 and EPA-HQ-OAR-2018-0794-1125.

[7] Ten EGUs at six plants were in operation when the agency took comment on this action. Since that time, four EGUs at two plants have indicated changes in operating status such that the units will be unaffected by this rule. Two units are in Pennsylvania (Cambria Cogen), and two units are in West Virginia (Morgantown). See preamble section III for more discussion of the operating status of these units.

EGUs, ranging from 40 megawatts (MW) to 110 MW. At the plant level, capacities range from 50 MW to 110 MW. The annual average SO_2 emissions rates vary across units as well, ranging from 0.29 to 0.56 pounds per MMBtu (lb/MMBtu) based on 2016 data.[8] We focus on SO_2 emissions rather than HCl or other acid gas HAP because these units have reported SO_2 emissions to the EPA but not emissions of other acid gas pollutants.[9] Stack heights at these units range from 250 to 400 feet.

Table 1. Eastern Bituminous Coal Refuse-fired EGUs in subcategory

Plant	State	ORIS Plant Code	2016 Net Summer Capacity (MW)[a]	2016 Annual Average SO_2 Emissions Rate (lb/MMBtu)[b]	Stack Hight (feet)
Colver Power Project	PA	10143	110	0.51	400
Ebensburg Power	PA	10603	50	0.56	250
Grant Town Power Plant Unit 1A	WV	10151	40	0.48	327[c]
Grant Town Power Plant Unit 1B	WV	10151	40	0.48	
Scrubgrass Generating Company LP Unit 1	PA	50974	42	0.29	363
Scrubgrass Generating Company LP Unit 2	PA	50974	42	0.31	363

[a] Source: *https://www.eia.gov/electricity/data/eia860/*.
[b] Source: *https://ampd.epa.gov/ampd/*.
[c] The two units at Grant Town Power Plant vent to a single stack.

We focus the compliance cost and emissions analysis in the year 2016 because EBCR-fired unit emissions from 2016 are used in the air quality modeling performed for this analysis. While other years may be used as a base year for the analysis, net generation, heat input, and SO_2 emissions levels in 2015, 2017, and 2018 were similar to levels in 2016, as is shown in Tables 2-4.

Table 2. Plant-level total Net Generation (MWh) for Eastern Bituminous Coal Refuse-fired EGUs in subcategory, 2015 to 2018[a]

Plant	State	2015	2016	2017	2018
Colver Power Project	PA	829,279	730,857	811,449	812,022
Ebensburg Power	PA	196,980	195,099	249,137	325,905
Grant Town Power Plant	WV	515,588	672,244	661,386	632,907
Scrubgrass Generating Company LP	PA	259,422	417,030	433,902	425,306
Total		**1,801,269**	**2,015,230**	**2,155,874**	**2,196,140**

[a] Source: *https://www.eia.gov/electricity/data/eia923/*.

[8] SO_2 emissions rates can be affected by a variety of factors, including the unit's efficiency in converting heat into electricity, the sulfur content of the input fuel, and the use of environmental control technology (*e.g.*, flue gas desulfurization (FGD) and sorbent injection technology).
[9] Sources may demonstrate compliance with the MATS acid gas limits by submitting measured emissions of either SO_2 or HCl. Most sources have chosen to submit SO_2 emissions data as they are already measuring and submitting that data to EPA. It is reasonable to assume the EBCR units may do the same.

Table 3. Total Heat Input (MMBtu) for Eastern Bituminous Coal Refuse-fired EGUs in subcategory, 2015 to 2018[a]

Plant	State	2015	2016	2017	2018
Colver Power Project	PA	10,413,109	9,312,141	10,256,988	10,393,739
Ebensburg Power	PA	3,014,189	3,199,391	4,131,375	5,590,563
Grant Town Power Plant Unit 1A	WV	3,869,732	4,978,923	4,608,821	4,624,559
Grant Town Power Plant Unit 1B	WV	4,129,354	4,878,864	4,788,024	4,606,781
Scrubgrass Generating Company LP Unit 1	PA	2,093,463	3,800,404	3,613,168	3,259,319
Scrubgrass Generating Company LP Unit 2	PA	2,150,215	3,553,787	3,372,466	3,267,768
Total		**25,670,062**	**29,723,510**	**30,770,842**	**31,742,729**

[a] Source: *https://ampd.epa.gov/ampd/*.

Table 4. Total SO_2 Emissions (short tons) for Eastern Bituminous Coal Refuse-fired EGUs in subcategory, 2015 to 2018[a]

Plant	State	2015	2016	2017	2018
Colver Power Project	PA	2,604	2,385	2,630	2,725
Ebensburg Power	PA	962	895	1,180	1,855
Grant Town Power Plant Unit 1A	WV	840	1,197	876	838
Grant Town Power Plant Unit 1B	WV	894	1,173	905	846
Scrubgrass Generating Company LP Unit 1	PA	354	558	526	412
Scrubgrass Generating Company LP Unit 2	PA	370	559	510	463
Total		**6,024**	**6,767**	**6,627**	**7,139**

[a] Source: *https://ampd.epa.gov/ampd/*.

2. Compliance cost reductions and forgone emissions reductions

In this section, we present estimates for the compliance cost reductions and forgone emissions reductions associated with the subcategorization for the representative year of analysis 2023. We estimate compliance cost reductions and forgone emissions reductions in 2023 based on the assumption that all units operate consistent with 2016 generation levels and install and operate dry scrubbers. While we assume in the primary baseline for this analysis that units continue to operate and comply with the acid gas limits in the 2012 MATS rule, we recognize that the dry scrubbers assumed in the analysis to be needed to comply with the acid gas limits have not been installed at these units. Therefore, the compliance cost reduction estimates under the primary baseline include capital cost reductions associated with not installing dry scrubbers as well as any associated reductions in variable costs. As is presented in Table 6 below, on an annualized basis, avoided capital expenditures are approximately 80 percent of the total avoided costs.

Because these units collectively represent a very small overall capacity, installation of emissions controls would not likely have a significant impact on dispatch. Therefore, this analytical approach is appropriate and sufficient for estimating the potential costs and benefits of this rule in a representative year.

In our primary baseline, we assume that all units install dry scrubbers and comply with the 2012 MATS-mandated SO_2 emissions rate limit of 0.2 lb/MMBtu. In the policy case of the primary scenario, we assume all units emit at their observed 2016 SO_2 emissions rates as shown in Table 1 and thus meet the subcategory SO_2 emissions limit of 0.6 lb/MMBtu without controls in addition to those used in 2016. Our compliance assumption in the primary baseline is consistent with the assumption made in the 2012 MATS RIA and is an approach to compliance for which the EPA is able to estimate the cost.[10] While commenters submitted information regarding other control alternatives that appear to be lower in cost, the commenters did not state that these costs were representative of all sources in this subcategory and also noted that even on their own unit, there were a number of reasons the control options might not be technically feasible, and therefore this information was insufficient to support the development of a compliance cost reductions estimate that is applicable to all sources in this subcategory.[11]

a. Compliance cost analysis

Under the primary scenario, we assume units will not incur additional 2012 MATS compliance costs as a result of the subcategorization. Rather than rely upon cost estimates for these units from analysis accompanying the 2012 MATS rule, we provide updated estimates of the compliance costs and emissions reductions associated with dry scrubbers, using unit-specific data and the EPA's Retrofit Cost Analyzer (RCA) tool.[12]

The RCA tool input data were obtained from a variety of publicly-available sources, including the EPA, the U.S. Energy Information Administration (EIA), and the U.S. Geological Survey (USGS) (Table 5). Except for capacity, which is provided in Table 1, other RCA inputs are set to default values.

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[10] Other post-combustion control options considered by EPA included dry sorbent injection (DSI) and wet FGD. Industry commenters expressed concern that DSI would render the ash produced as a by-product of combustion unusable in reclamation activities due to high alkalinity. In this case, the commenters claimed that the ash would need to be landfilled, which would increase costs significantly due to landfill tipping fees. Alternatively, industry input and EPA technical judgment suggested that wet FGD is a more costly compliance option than dry scrubbers. It could not be determined whether it would be less costly to comply with the HCl emission rate rather than SO_2 emission rate because of limited information on the HCl emissions from these units. Another potential control strategy is to increase the amount of limestone that is injected into these fluidized bed combustors (FBC). Commenters asserted that additional limestone injected can result in a decrease in the electric generation load and amount of coal refuse that is injected (as there is a limit to the quantity of solids that can be injected). For these reasons, EPA assumes in the baseline that units would use traditional dry scrubbers to comply with the current acid gas standards.

[11] See Docket ID No. EPA-HQ-OAR-2018-0794-1260.

[12] *https://www.epa.gov/airmarkets/retrofit-cost-analyzer*.

Table 5. EGU input variables and data sources

Input Variable	Source	Data Year
Generation levels	EIA Form 923[13]	2016
SO_2 emissions levels	EPA Air Markets Program Data (AMPD)[14]	2016
Heat input levels	EPA AMPD	2016
Heat rate	EPA National Electric Energy Data System (NEEDS) v6[15]	N/A
Lime price	USGS Mineral Commodity Summaries[16]	2018

Detailed compliance cost results from the RCA tool are provided in Table 6. For the plants with multiple generating units, we assume that, due to the small size of the units (less than 50 MW each), one dry scrubber is installed for the entire plant. Upfront capital costs are annualized, assuming a 14.05 percent capital charge rate. The capital charge is based on the EPA's current modeling assumption for merchant-owned environmental retrofits. The capital charge rate is a function of the following parameters: capital structure (debt/equity shares of an investment), pre-tax debt rate, debt life, the post-tax return on equity, and other costs such as property taxes and insurance, state and federal corporate income taxes, depreciation schedule, and book life.[17] As a result, the compliance cost reductions presented in this section are a measure of the reduction in private expenditures for affected EBCR-fired units under the primary baseline due to the subcategorization. As the cost reduction estimates include reductions in state and federal taxes via the application of the capital charge rate, which represent transfer payments, the estimates in this section do not serve as a measure of the change in social costs due to the subcategorization; estimates of social cost are provided in net benefits analysis in Section 5 of this memo using 3 and 7 percent discount rates consistent with OMB guidance.

Based on the dry scrubbing estimates from the RCA tool, we estimate that installation of dry scrubbers would require annual expenditures of $49.2 million for the four plants.[18] The bulk of the estimated annual expenditures ($39.3 million) account for annualized payments for capital, with total upfront capital expenditures of about $280 million for 324 MW of capacity. At the plant level, estimated annualized expenditures range from $9.4 million at the smallest plant (Ebensburg) to $14.3 million at the largest plant (Colver).

[13] *https://www.eia.gov/electricity/data/eia923/*.
[14] *https://ampd.epa.gov/ampd/*.
[15] *https://www.epa.gov/airmarkets/national-electric-energy-data-system-needs-v6*. Heat input levels in NEEDS are based on the 2017 Annual Energy Outlook, which is a reasonable approximation of 2016 observed levels,
[16] *https://prd-wret.s3-us-west-2.amazonaws.com/assets/palladium/production/atoms/files/mcs-2019-lime.pdf*.
[17] For more information, see Chapter 10 and Table 10-10 of Documentation for EPA Power Sector Modeling Platform v6 November 2018 Reference Case. Access at: *https://www.epa.gov/airmarkets/documentation-epas-power-sector-modeling-platform-v6*.
[18] This calculation assumes an annualization period of 15 years, which is chosen for consistency with the assumption that capital investment would be financed over that period.

Table 6. Plant-level estimates of annual incremental compliance cost reductions in 2023 resulting from subcategorization, based on EPA Retrofit Cost Analyzer (2016$)[a]

Plant	State	Summer Capacity (MW)	Total Capital Expenditures ($M)	Annual Capital Expenditures ($M)	Annual O&M Expenditures ($M)	Total Annual Expenditures ($M)
Colver Power Project	PA	110	81.8	11.5	2.9	14.3
Ebensburg Power	PA	50	53.1	7.5	1.9	9.4
Grant Town Power Plant (Units 1A and 1B)	WV	80	71.3	10.0	2.8	12.9
Scrubgrass Generating Company LP (Units 1 and 2)	PA	84	73.4	10.3	2.3	12.6
Total	**---**	**324**	**279.6**	**39.3**	**9.9**	**49.2**

[a] The compliance cost reductions presented in this section, which are total annual expenditures, are a measure of the reduction in private expenditures for affected EBCR-fired units under the primary baseline due to the subcategorization. The total annual expenditures are inclusive of state and federal taxes that are accounted for in the capital charge rate applied to total capital expenditures.

As a result of the subcategorization, we anticipate forgone reductions of acid gas emissions. These forgone reductions include acid gases that are considered HAP under section 112 of the Clean Air Act, as well as SO_2, which is an acid gas but not a HAP. Due to data limitations, we are unable to quantify forgone emissions reductions from acid gas HAP other than HCl. While other acid gas pollutants are expected to change in a similar manner as HCl and SO_2, emissions of non-acid gas pollutants are not expected to change significantly between the primary baseline and policy case.

For HCl and SO_2, we compare emission estimates between the policy case and the primary baseline (the 2012 MATS limit). For HCl, we estimate forgone emissions reductions based on the difference between the 2012 MATS emissions rate limits and the limits established by the subcategorization, assuming 2016 fuel use levels. For SO_2, we leverage 2016 unit-specific measurement data and the 2012 MATS emissions rate limits to estimate forgone emissions reductions.

Annual unit-level HCl forgone emissions reductions estimates are provided in Table 7. Since measured HCl emissions data are unavailable, we assume that units emit at the HCl emissions rate limit established by the subcategorization (0.04 lb/MMBtu) in the policy case, and at the 2012 MATS HCl emissions rate limit (0.002 lb/MMBtu) in the primary baseline. In both cases, we assume the same fuel use as was reported for each unit in 2016. These assumptions result in an annual HCl emissions estimate of 594 tons in the policy case and 30 tons in the primary baseline. Therefore, we estimate that the subcategorization will result in 565 tons of forgone HCl emissions reductions annually.

Table 7. Estimates of annual unit-level HCl forgone emissions reductions in 2023 resulting from subcategorization under the primary baseline

Plant	State	2016 Heat Input (MMBtu)	Assumed HCl Emissions at Subcategory Limit of 0.04 lb/MMBtu (short tons)	Assumed HCl Emissions at Baseline 2012 MATS Limit of 0.002 lb/MMBtu (short tons)	Forgone HCl Emissions Reductions (short tons)
Colver Power Project	PA	9,312,141	186	9	177
Ebensburg Power	PA	3,199,391	64	3	61
Grant Town Power Plant Unit 1A	WV	4,978,923	100	5	95
Grant Town Power Plant Unit 1B	WV	4,878,864	98	5	93
Scrubgrass Generating Company LP Unit 1	PA	3,800,404	76	4	72
Scrubgrass Generating Company LP Unit 2	PA	3,553,787	71	4	68
Total	**---**	**29,723,510**	**594**	**30**	**565**

Annual unit-level SO_2 forgone emissions reductions estimates are provided in Table 8. The six units in this subcategory reported a total of 6,767 tons of SO_2 emissions in 2016 to the EPA. Since we assume future operating conditions identical to those observed in 2016, we estimate the SO_2 emissions levels in the policy case are also 6,767 tons annually. For the primary baseline, we assume that each unit emits at the 2012 MATS limit of 0.2 lb/MMBtu and uses the same amount of fuel as in 2016, resulting in an annual estimate of 2,972 tons of SO_2. We therefore estimate that the subcategorization results in total forgone SO_2 emissions reductions of 3,794 tons annually: 1,384 tons from the West Virginia plants and 2,410 tons from the Pennsylvania plants.

Table 8. Estimates of annual unit-level SO_2 forgone emissions reductions in 2023 resulting from subcategorization under primary baseline

Plant	State	2016 Heat Input (MMBtu)	2016 SO_2 Emissions (short tons)	Assumed SO_2 Emissions at Baseline 2012 MATS Limit of 0.2 lb/MMBtu (short tons)	Forgone SO_2 Emissions Reductions (short tons)
Colver Power Project	PA	9,312,141	2,385	931	1,453
Ebensburg Power	PA	3,199,391	895	320	575
Grant Town Power Plant Unit 1A	WV	4,978,923	1,197	498	699
Grant Town Power Plant Unit 1B	WV	4,878,864	1,173	488	685
Scrubgrass Generating Company LP Unit 1	PA	3,800,404	558	380	178
Scrubgrass Generating Company LP Unit 2	PA	3,553,787	559	355	203
Total	**---**	**29,723,510**	**6,767**	**2,972**	**3,794**

3. Forgone benefits

a. Forgone targeted HAP Benefits

The affected EBCR-fired units emit acid gas HAP, including HCl, HF, HCN, and SeO_2. Limited data and methods prevent us from quantifying the economic value of the forgone HAP emission reductions that may result from this subcategorization.

b. Forgone $PM_{2.5}$-related co-benefits

MATS compliance strategies reduce HAP emissions but also influence the emissions of other pollutants that adversely affect human health, particularly the criteria pollutant SO_2, a precursor to ambient $PM_{2.5}$. The change in emissions between the baseline and policy cases shown in Table 8 may affect ambient concentrations, population exposure, and human health impacts associated with $PM_{2.5}$. For the purposes of this memo, forgone benefits associated with forgone SO_2 emissions reductions are considered "forgone co-benefits", as SO_2 is not a targeted pollutant in this rulemaking.

In this section of the memo, we report the estimated number and economic value of forgone $PM_{2.5}$-attributable premature deaths and illnesses. The analysis to quantify forgone co-benefits from projected changes in $PM_{2.5}$ concentrations was initially conducted using baseline and policy scenarios that included ten ECBR-fired units that at the time were expected to be affected by this action. All units were assumed to install dry scrubbers in the primary baseline and to remain in operation in the primary baseline and policy case, consistent with the assumptions described above. As mentioned above (in footnote 6), four of those EBCR-fired units (two units in a single Pennsylvania plant and two units in a single West Virginia plant) have changed operating status and will not be affected by this action. The remaining six units are those identified in Table 1. The analysis of the ten ECBR-fired units included detailed air quality modeling and benefits analyses described below. The forgone SO_2 emissions reductions shown in Table 8 are 65 percent of the forgone SO_2 reductions that were estimated in the initial analysis based on the larger set of ten units. To approximate the impacts of this final action for the six units shown in Table 8, we scale the estimates of incidence and forgone benefits of the initial analysis using ten ECBR-fired units by 0.65 (*i.e.*, we multiply the results of the initial analysis by 0.65).

We believe the forgone benefits estimate is a reasonable approximation for the purpose of this rule for two reasons. First, the nature of sulfate as a secondary pollutant and forgone SO_2 emissions being in a localized geographic region mean that the footprint of the impacts is not expected to change substantially due to change in operating status for the Cambria Cogen and Morgantown units. Second, the annual average $PM_{2.5}$ concentrations across the impacted region[19] are expected to change by less than 0.1 $\mu g/m^3$ due to the change on operating status of these units, meaning that the original analysis showing the distribution of mortality effects occurring at different concentrations (*e.g.*, cutpoints) can be reasonably scaled without

[19] The locations seeing the largest impacts all have $PM_{2.5}$ concentrations less than 12 $\mu g/m^3$ in the baseline scenario.

accounting for changes in baseline $PM_{2.5}$ concentrations. These issues are discussed in more detail in Section 6.

In the section below, we describe our approach to simulating the change in annual mean $PM_{2.5}$ estimated to result from subcategorizing the ten EBCR-fired units operating when we performed the air quality analysis. Next, we detail briefly our methods for estimating the forgone $PM_{2.5}$-related co-benefits. We then report the estimated forgone benefits in the form of forgone $PM_{2.5}$-attributable deaths and illnesses avoided and the forgone economic value of these attributable deaths and illnesses.

i. Simulating the change in annual mean $PM_{2.5}$ estimated to result from subcategorization

Creation of annual $PM_{2.5}$ spatial fields representing the primary baseline and policy cases leveraged available photochemical modeling outputs that were created as part of the Regulatory Impact Analysis for the Repeal of the Clean Power Plan, and the Emission Guidelines for Greenhouse Gas Emissions from Existing Electric Utility Generating Units (U.S. EPA 2019b), also referred to the Affordable Clean Energy (ACE) rule. These $PM_{2.5}$ spatial fields were used as input to BenMAP-CE, which, in turn, was used to quantify the forgone benefits from this final rule.

The analysis supporting this rule used outputs from several full-scale photochemical model simulations. When possible, the EPA utilizes full-scale modeling to estimate potential impacts from regulatory actions. Full-scale modeling simulating pollutant concentrations for the specific sector, analytical year and regulatory scenario is more accurate than a reduced-form approach. In cases where full-scale modeling is not feasible, the EPA has often employed a reduced-form "benefit per ton" (BPT) approach. A BPT approach is not being used in this analysis. Reduced form tools are less complex than full-scale air quality modeling, requiring less agency resources and time. The EPA is currently working on a systematic comparison of results from its BPT technique and other reduced-form techniques with results from full-form photochemical modelling.

While this analysis employed photochemical modeling simulations, we acknowledge that the Agency has elsewhere applied reduced-form techniques. The summary report from the "Reduced Form Tool Evaluation Project", which has not yet been peer reviewed, is available on the EPA's website at *https://www.epa.gov/benmap/reduced-form-evaluation-project-report*. Under the scenarios examined in that report, the EPA's 2012 BPT approach (which was based off a 2005 inventory) may yield estimates of $PM_{2.5}$- benefits for the EGU sector that are as much as 30 percent greater than those estimated when using full air quality modeling. The EPA continues to work to develop refined reduced-form approaches for estimating $PM_{2.5}$ benefits.

The full-scale modeling used in this analysis included annual model simulations for a 2011 base year and a 2023 future year to provide hourly concentrations of primary and secondarily formed $PM_{2.5}$ component species (*e.g.*, sulfate, nitrate, ammonium, elemental carbon, organic matter, and crustal material) for both years nationwide. For the EGUs in this subcategory, emissions used in the 2023 modeled future year are equal to the actual 2016 emissions and are consistent with the policy case described above for the six plants included in Table 8 but do not represent the recent changes in operating status of the Cambria Cogen and Morgantown facilities. Emissions in the 2023 primary baseline and policy case from other source categories represent projections of emissions levels from these sources based on air pollution regulations that were on-the-books at the time the modeling was conducted as well as projections of population growth, energy demand and other factors which impact emissions (U.S. EPA 2017a).

As described in more detail in the appendix to this memo, the photochemical modeling results for 2011 and 2023, in conjunction with modeling to characterize the air quality impacts from groups of emissions sources (*i.e.*, source apportionment modeling) and emissions data for the primary baseline and policy case, were used to construct the air quality spatial fields that reflect the influence of EBCR-fired EGU SO_2 emissions from the ten ECBR-fired units in the initial analysis on $PM_{2.5}$ concentrations for the primary baseline and policy case. While emissions from other sectors represent a 2023 future year case, those emission are held constant between the baseline and policy case, so changes in $PM_{2.5}$ between the cases reflect only the impact of the policy on EBCR-fired EGUs.

The photochemical model simulations as well as the basic methodology for determining air quality changes are the same as those used in the ACE RIA. The appendix to this memo provides an overview of the air quality modeling and the methodologies we used to develop spatial fields of annual $PM_{2.5}$ concentrations. Additional information on the air quality modeling platform (inputs and set-up), model performance evaluation for $PM_{2.5}$, emissions processing for this analysis, and additional details and numerical examples of the methodologies for developing $PM_{2.5}$ spatial fields are available (U.S. EPA 2019b, chap. 8).

ii. Approach to estimating forgone $PM_{2.5}$-related health impacts

Using the open-source environmental Benefits Mapping and Analysis Program—Community Edition (BenMAP-CE) (Sacks et al. 2018), we estimate the forgone co-benefits associated with the subcategorization. The procedure for calculating and valuing air pollution-related impacts is described in detail elsewhere (U.S. EPA 2012b; Fann et al. 2018; Sacks et al. 2018), and so we briefly summarize the approach here.

The BenMAP-CE tool uses a health impact function to quantify excess cases of air pollution-attributable premature deaths and illnesses. When used to quantify $PM_{2.5}$-related effects, the function combines an effect estimate (*i.e.*, the β coefficient) from an epidemiological study, which portrays the relationship between a change in air quality and a health effect, such as mortality, with estimated $PM_{2.5}$ concentrations (supplied using the model simulations described above), population data, and baseline death rates for each county in each year.

After having quantified $PM_{2.5}$-attributable cases of premature death and illness, we estimate the economic value of these values using willingness-to-pay and cost-of-illness measures. For this analysis we applied version 1.5 of the tool (U.S. EPA 2019a). The Appendix to the BenMAP-CE user manual and the RIA for the Particulate Matter National Ambient Air Quality Standards each detail the source of the above input parameters (U.S. EPA 2012a; 2018).

We estimate the number of $PM_{2.5}$-attributable premature deaths using effect estimates from two epidemiology studies examining two large population cohorts: the American Cancer Society (Krewski et al. 2009) and the Harvard Six Cities (Lepeule et al. 2012) cohorts. Consistent with the ACE RIA (U.S. EPA 2019b), we report the estimated number of $PM_{2.5}$-attributable deaths according to alternative $PM_{2.5}$ concentration cutpoints. This approach allows readers to determine the portion of the population exposed to annual mean $PM_{2.5}$ levels at or above different concentrations. The Agency does not view these concentration cutpoints as thresholds below which we would not quantify the human health impacts attributable to $PM_{2.5}$.

iii. Air quality and health benefit analysis results

Changes in annual mean $PM_{2.5}$ concentrations between the baseline and the policy case for the ten ECBR-fired units in the initial analysis are shown in the appendix (Figure 5). The largest changes in $PM_{2.5}$ concentrations are estimated to occur in Pennsylvania and West Virginia. The spatial patterns of predicted impacts are a result of (1) of the spatial distribution of sources and emissions for the four source apportionment "tags"[20] which contain EBCR-fired units and (2) of the physical or chemical processing that the model simulates in the atmosphere. The spatial fields of predicted $PM_{2.5}$ impacts serve as inputs to the benefits analysis, the results of which are described below.

Below we report the forgone number of $PM_{2.5}$-related premature deaths and illnesses estimated to occur in 2023 (Table 9). We next report the number of $PM_{2.5}$-attributable premature deaths estimated to occur at alternative concentration cutpoints including the Lowest Measured Level of each long-term epidemiologic study and the $PM_{2.5}$ annual mean NAAQS (Table 10); most premature deaths are estimated to occur below the level of the PM NAAQS. Finally, we report the estimated economic value of the forgone $PM_{2.5}$-attributable premature deaths and illnesses in 2023 (Table 11). Note that these values have been scaled by 65 percent to account for the removal of the Cambria Cogen and Morgantown units.

[20] "Tags" refer to groups of sources whose air quality impacts were tracked in tandem. The method is described in detail in the appendix and in EPA (2019b), which explain that impacts from tagged groups of sources were scaled up or down based on overall emissions changes for each tag between the baseline and policy case. The method does not account for any changes in spatial distribution of emissions within the tag.

Table 9. Estimated number of $PM_{2.5}$-attributable premature deaths and illnesses in 2023

Health Endpoint	Forgone $PM_{2.5}$-Attributable Health Impacts (95% confidence interval)[a]
Premature deaths	
Krewski et al. (2009)	24 (16 to 32)
Lepeule et al. (2012)	55 (27 to 82)
$PM_{2.5}$-related non-fatal heart attacks	
Peters et al. (2001)	24 (5.8 to 41)
Pooled estimate	2.5 (0.94 to 6.8)
All other morbidity effects	
Hospital admissions—cardiovascular	6.0 (4.9 to 8.8)
Hospital admissions—respiratory	5.8 (0.17 to 9.1)
Emergency department visits for asthma	15 (-5.7 to 32)
Exacerbated asthma	720 (-42 to 1,600)
Minor restricted activity days	17,000 (14,000 to 20,000)
Acute bronchitis	29 (-6.8 to 65)
Upper respiratory symptoms	530 (95 to 950)
Lower respiratory symptoms	370 (140 to 600)
Lost work days	2,800 (2,400 to 3,200)

[a] Values rounded to two significant figures.

Table 10. Estimated number of $PM_{2.5}$-attributable premature deaths occurring above and below concentration cutpoints in 2023

Epidemiologic Study	Total $PM_{2.5}$-attributable deaths	$PM_{2.5}$-attributable deaths reported by air quality cutpoint		
		Above NAAQS	*Below NAAQS and Above LML*[a]	*Below LML*
Krewski et al. (2009)	24	<1	20	4
Lepeule et al. (2012)	55	<1	15	40

[a] The LML of the Krewski et al. (2009) study is 5.8 µg/m³. The LML of the Lepeule et al. (2012) study is 8 µg/m³.

Table 11. Estimated economic value of the forgone $PM_{2.5}$-attributable premature deaths and illnesses in 2023

Approach to Estimating the Value of $PM_{2.5}$-Related Premature Deaths	Estimated Value of $PM_{2.5}$-Related Premature Deaths (millions of 2016$, value of mortality and morbidity effects)[a]
Benefits discounted at 3%	
No-threshold model	$230 to $530
Limited to above LML	$150 to $200
Effects above NAAQS	$4.0 to $7.2
Benefits discounted at 7%	
No-threshold model	$210 to $480
Limited to above LML	$140 to $180
Effects above NAAQS	$4.0 to $7.1

[a] Low end of range reflects dollar value of effects quantified using the concentration-response parameter from Krewski et al. (2009), and the upper end is quantified using the parameter from Lepeule et al. (2012).

4. Impacts under an alternative baseline

In the preceding cost, emissions, and benefits analysis, we assumed that, absent the subcategorization, each EGU would comply with the 2012 MATS HCl emissions rate limit by installing dry scrubber control technology. However, it is possible that rather than installing these controls to comply, some or all EGUs would be retired instead in the baseline, and that these retirements otherwise would not occur in the policy case. In this instance, compliance cost reductions and forgone benefits would be different, and other environmental and economic impacts may result. While we lack information to estimate compliance cost reductions and forgone benefits under a baseline in which units retire but otherwise would not do so in the policy case, we provide a qualitative discussion of the relevant impacts of such an outcome.

a. Compliance cost reductions

If we assume that units were to retire rather than install controls in the baseline, then our avoided cost estimate would no longer include the cost of installing and operating those controls. Instead, compliance cost reductions would reflect the additional generation costs from non-EBCR-fired EGUs required to replace the production from the retiring units minus generation costs that would occur at the EBCR-fired EGUs in the policy case. Since the units in the subcategory represent a small fraction of capacity in their respective service areas,[21] we anticipate that the loss of their generation would be offset, at least in the short run, by increased generation from other EGUs, likely natural gas- and other coal-fired units. Under the alternative baseline and assumptions of competitive markets, profit-maximizing behavior by generators, and approximately inelastic demand for electricity, we would expect the avoided additional generation costs from non-EBCR-fired EGUs in the policy case to be less than the compliance cost reductions of installing controls. That is, if a unit retires in the baseline but not in the policy case, it is because its profits from complying in the baseline are less than the cost of installing the controls, which implies that the costs must be lower if they retire than if they install the controls. Therefore, under a baseline in which some or all EBCR-fired EGUs retire, we would expect compliance cost reductions to be less than estimated in Table 7 and Table 8.

It is difficult to quantify the impact that the additional costs of control will have on the operation of waste coal facilities in part because they operate under a variety of different cost structures, and the EPA has limited information about the specifics of the cost structure each unit is operating under. Some are likely to be operating under the PJM market structure where decisions would be made based on projections of the likely future wholesale and capacity prices in PJM. Others are likely operating under Public Utility Regulatory Policies Act of 1978 (PURPA) contracts. For these units, decisions would be made based on how the additional costs relate to the price under the PURPA contracts (which very well may be higher than wholesale PJM prices). For these units, the question is further complicated by the fact that the PURPA contracts are not indefinite and the year in which they expire could create even greater uncertainty about future expected returns on investment for those plants. While the EPA cannot fully analyze this situation, ESI (2019) presents information demonstrating the market challenges faced by EBCR-fired units in the competitive PJM. These challenges suggest that the increased cost of MATS compliance could impact utilization and make the qualitative impacts discussed in this section important to consider.

[21] The 2022-23 resource model from PJM, the regional transmission organization (RTO) whose territory includes the units in the subcategory, indicates installed capacity of 8,141 MW and 8,688 MW in the APS and PENELEC zones, respectively. The APS zone includes the West Virginia units in the subcategory, which the PENELEC zone includes the Pennsylvania units. The resource model can be accessed for past and future years at *https://www.pjm.com/markets-and-operations/rpm.aspx*.

b. Forgone emissions reductions and forgone air quality benefits

Likewise, we expect forgone acid gas HAP and SO_2 emissions reductions to be different if units were to retire rather than install controls in the baseline. Unlike compliance cost reductions, however, we anticipate that forgone emissions benefits, both forgone targeted HAP benefits and co-benefits, would likely be greater than estimated in Table 7 and Table 8, with the degree depending on the characteristics of the non-EBCR-fired EGUs providing generation in the baseline that is displaced by the increased generation of EBCR-fired EGUs in the policy case. For example, if generation in the baseline displaced in the policy case is exclusively natural gas-fired generation, then the increased acid gas emissions from this rule would be greater in this alternative baseline as natural gas generation has a lower acid gas emission rate than the MATS standard for coal-fired EGUs. On the other hand, if baseline generation displaced in the policy case was provided exclusively by coal-fired EGUs complying with the 2012 MATS emission rate limits, then the change in acid gas emissions from this final rule would be roughly equivalent across the two baselines (because the primary baseline assumes that the EBCR-fired EGUs would operate and emit at the 2012 MATS emission rate limits). More likely than either extreme outcome, however, would be that the generation displaced by the EBCR-fired EGUs in the policy case was provided by a mix of natural gas- and coal-fired units in the baseline. In this likely case the avoided emission reductions from the subcategorization would be greater in the alternative baseline relative to the primary baseline. In turn, we expect that the forgone unquantified health benefits (from acid gas HAP emission reductions) and quantified forgone health co-benefits (from SO_2 emission reductions) would be greater under the alternative baseline, although this conclusion also depends on the location of the non-EBCR-fired generation in the baseline that is displaced in the policy case because SO_2 damages depend on the location of emissions. Furthermore, emissions of pollutants other than acid gases (*e.g.*, nitrogen oxides) would be different between the baseline and policy case depending on the emission rates of the displaced power relative to the emission rates of the EBCR-fired EGUs.

c. Forgone air and water quality-related benefits from coal refuse remediation

Unlike the baseline in which all EBCR-fired EGUs install controls, there are additional benefits of remediating coal refuse piles provided by these EGUs that would be higher in the policy case relative to the alternative baseline where some of these of units retire. These are co-benefits from the policy case relative to this alternative baseline as they are benefits that are not from targeted pollutants in this rulemaking (*i.e.*, acid gas air emissions from EBCR-fired EGUs) but, while difficult to quantify, they may be significant and should be accounted for in a benefit-cost analysis. As the EBCR-fired units obtain their fuel regionally from coal refuse piles, if some, but not all, units retire, the units remaining in operation may consume some of the coal refuse that would have been consumed had the retired unit(s) not retired, potentially changing the time path of the consumption of the coal refuse piles.

Reports by the Pennsylvania Department of Health and the U.S. Department of the Interior (DOI) discussed how coal refuse piles found in abandoned mine lands (AMLs) can be ignited through spontaneous combustion or human activity (Sussman and Mulhern 1964; McNay 1971).

The unmanaged combustion of coal refuse piles can release harmful pollutants in an uncontrolled manner, including acid gases and other HAP, into the surrounding environment. The burning coal refuse piles may require actions from area fire departments, meaning that the removal of the piles may also avert expenditures on fire services. In the absence of mitigation efforts, refuse piles may continue to smolder for decades (Sussman and Mulhern 1964).

Several studies attempt to quantify the emissions impacts of burning coal refuse piles through site-level sampling techniques. Chalekode and Blackwell (1978), in a report prepared for the EPA, presented estimates of the emissions of a variety of air pollutants from a burning bituminous coal refuse pile, which they deem to be representative of nearby sources. The authors report emissions factors for total particulates, nitrogen oxides, sulfur oxides, hydrocarbons, carbon monoxide, hydrogen sulfide, ammonia, mercury, and polycyclic organic materials. The emissions factor for sulfur oxides is 7.4×10^{-5} kilograms per hour per metric ton of burning refuse. A later DOI report estimates emissions at a burning refuse pile near Albright, WV (Chaiken and Bayles 1991). Based on the material composition of the pile samples, the authors estimate an upper bound of 7.1 lbs. of SO_2 per MMBtu of refuse, which is almost twelve times larger than the subcategory SO_2 emissions rate standard.

Unfortunately, the existing literature does not contain enough information to provide a meaningful quantitative assessment of the emissions impacts of the policy under the alternative baseline. Even if we knew with certainty that every ton of coal refuse consumed at the EBCR-fired EGUs in a given year would otherwise burn uncontrolled, which is an extreme assumption, we would still need to know the rate at which that refuse would burn (*e.g.*, in terms of MMBtu per year) in the uncontrolled setting in order to directly compare emissions between burning the coal in the landscape or in a EBCR-fired EGG. Furthermore, since we know that only a subset of piles are burning at any point in time, we would also need an estimate of the hazard rate for uncontrolled burning of the piles. Moreover, we would need to identify the probability that a pile that is burning uncontrollably would be consumed by an EBCR-fired EGU.

Short of a quantitative analysis, we still recognize the remediation benefit that the EBCR-fired EGUs provide with respect to air quality. If the subcategorization prevents EBCR-fired EGUs from retiring, and instead results in coal refuse piles that would have otherwise ignited at the mine site being used as fuel,[22] then the forgone emissions reductions estimated in Table 7 and Table 8 would overestimate the air quality impacts of the subcategorization.

Coal refuse piles can also create water quality issues via acid mine drainage (AMD). Pollutants found in AMD include acidity, metals, solids, and increased conductivity (U.S. EPA 2008). Conductivity is measured as an indicator pollutant of total dissolved solids (TDS) which includes bicarbonate, calcium, magnesium, and sulfate (U.S. EPA 1982). These pollutants can harm human health and have ecological impacts in waterbodies (U.S. EPA 2000). The continued

[22] Using data from the Pennsylvania Department of Environmental Protection (PA-DEP), ESI (2019) finds that of the 772 refuse piles in the PA-DEP inventory, 45 are currently burning uncontrolled. This includes both bituminous and anthracite piles.

consumption and remediation of coal refuse piles can lead to reduced water quality issues as well as averted water treatment and environmental management costs.[23]

However, it is possible that the piles could be mitigated in another fashion. In a report to the Appalachian Region Independent Power Producers Association (ARIPPA), ESI (2019) estimates, based on bids submitted to the Pennsylvania Department of Environmental Protection (PA-DEP) for a coal refuse pile mitigation project, that removal and disposal would cost between $11 and $33 per ton of refuse and remediation would cost between $20,000 and $23,000 per acre. In 2016, the units affected by the subcategorization consumed 2.8 million tons of coal refuse.[24] Based on the report estimates,[25] that amount translates to about $32 to $94 million in annual compliance cost reductions, less the cost of removal, delivery, and remediation to the EBCR-fired EGUs, including tax credit subsidies.[26] This estimate assumes that coal refuse is consumed at the same rate in future years as in the policy case, that all these units retire in the baseline, and that states would pursue removal, disposal, and remediation in lieu of the coal refuse being consumed by the affected EBCR-fired EGUs.

5. Social net benefits

This section presents a series of estimates of the annual monetized social costs, benefits, and net benefits of the subcategorization based upon the quantified analysis in the primary scenario presented above. Note that in reporting the benefits, costs, and net benefits in the following tables, we modify the relevant terminology. Benefits are equal to the compliance cost reductions, costs equal the forgone benefits, and ancillary costs equal the forgone co-benefits.

Section 2 presented estimates of the reduction in private expenditures for affected EBCR-fired units under the primary baseline due to the subcategorization. As those compliance cost reduction estimates included reductions in state and federal taxes and other factors, we re-estimate the costs in this section for the purpose of net benefits analysis to approximate the change in social costs anticipated due to the subcategorization. The estimates of social cost reductions in this section apply 3 and 7 percent discount rates (combined with a 15-year economic lifetime) to annualize upfront capital expenditures.

[23] ESI (2019) discusses benefits associated with water quality, public health and safety, and land value improvements from coal refuse remediation services, estimating an average benefit of about $37 million per year in Pennsylvania for remediation projects in eastern bituminous and anthracite regions. We present the ESI (2019) estimates for illustrative purposes. For example, the improvements in land value presented by the study include both the value of returning land dedicated to coal refuse to productive uses and to improvements in property values for nearby parcels; property value increases of parcels near to remediated coal refuse piles may capitalize benefits from other improvements, such as water quality improvements and reduced health and safety risks, leading to potential double-counting issues.

[24] Input data was compiled from EIA Form 923. See *https://www.eia.gov/electricity/data/eia923/*.

[25] ESI (2019) equates 8 million tons of coal refuse to 240 acres of remediated AML. We apply the same ratio (1 million tons of coal refuse to 30 acres of remediated AML) for the coal refuse consumption of the EGUs in the subcategory.

[26] According to ESI (2019), coal refuse-fired EGUs in Pennsylvania receive a tax credit of up to $4 per ton of refuse, with a total cap on credits of $10 million per year.

a. Net benefits associated with the targeted acid gas HAP and criteria co-pollutants

When considering whether a regulatory action is a potential welfare improvement (*i.e.*, potential Pareto improvement), it is necessary to consider all impacts of the action. Therefore, Table 12 provide the estimates of the benefits, costs, and net benefits of the primary scenario, inclusive of the forgone beneficial impacts from the SO_2 emission changes that are projected to accompany the changes in HCl and other acid gas HAP emissions. In these tables, the estimates for the forgone ancillary health co-benefits are derived using $PM_{2.5}$ log-linear concentration-response functions that quantify risk associated with the full range of $PM_{2.5}$ exposures experienced by the population.

Table 12. Summary of social costs and benefits (millions of 2016$) of the subcategorization in 2023, policy case compared to the primary baseline[a]

Description	Estimate (3% Discount Rate)	Estimate (7% Discount Rate)
Social benefits		
Cost reductions[b]	$33	$41
Social costs		
Forgone targeted benefits[c]	C	C
Forgone ancillary co-benefits	$230 to $530	$210 to $480
Social net benefits		
Cost reductions minus forgone targeted benefits and forgone co-benefits	-$200 to -$490 – C	-$170 to -$440 – C

[a] Figures in this table are presented rounded to two significant figures. Totals may not sum due to independent rounding.
[b] The cost reductions presented in this section are estimates of the reductions in the social cost of the rule, as opposed to the reduction in private expenditures for affected EBCR-fired units presented in Section 2 of the memo, which was estimated to be $49 million in 2023. On an annualized basis, about 80 percent of the compliance cost reductions is associated with avoided capital expenditures.
[c] C is the sum of all unquantified forgone targeted acid gas HAP benefits (or costs in this table).

b. Net benefits associated with targeted acid gas HAP

In the decision-making process, it is useful to consider the change in (forgone) benefits due to the targeted pollutant relative to the (reduction in) costs. In Table 13, we offer one perspective on the benefits and costs of this subcategorization by presenting a comparison of the beneficial impact associated with compliance cost reductions and the forgone benefits associated with forgone reductions of the targeted pollutant, acid gas HAP.

Table 13. Summary of social costs and benefits (millions of 2016$) of the subcategorization in 2023 associated with the targeted acid gas HAP, policy case compared to the primary baseline[a]

Description	Estimate (3% Discount Rate)	Estimate (7% Discount Rate)
Social benefits		
Cost reductions[b]	$33	$41
Social costs		
Forgone targeted benefits[c]	C	C
Net benefits of the targeted pollutant only		
Cost reductions minus forgone targeted benefits	$33 - C	$41 – C

[a] Figures in this table are presented rounded to two significant figures. Totals may not sum due to independent rounding.
[b] The cost reductions presented in this section are estimates of the reductions in the social cost of the rule, as opposed to the reduction in private expenditures for affected EBCR-fired units presented in Section 2 of the memo, which was estimated to be $49 million in 2023. On an annualized basis, about 80 percent of the compliance cost reductions is associated with avoided capital expenditures.
[c] C is the sum of all unquantified forgone targeted acid gas HAP benefits (or costs in this table).

c. Net benefits including air pollution co-benefits calculated according to sensitivity analysis assumptions

Table 14 and Table 15 report the estimated benefits, costs, and net benefits of the primary scenario according to different sensitivity analysis assumptions. These results reflect different assumptions regarding the relationship between $PM_{2.5}$ exposure and the risk of premature death. In Table 12, we report the net benefits calculated using estimates of the forgone $PM_{2.5}$-related benefits based on a no-threshold concentration-response parameter for $PM_{2.5}$. In Table 14, we report the net benefits calculated using estimates of the forgone $PM_{2.5}$-related co-benefits assuming that the $PM_{2.5}$-attributable risks fall to zero below the lowest measured levels of the two long-term $PM_{2.5}$ mortality studies used to quantify risk. In Table 15, we report the net benefits calculated using the estimated forgone $PM_{2.5}$-related benefits assuming that $PM_{2.5}$ related benefits fall to zero below the $PM_{2.5}$ National Ambient Air Quality Standard (NAAQS).

The EPA has generally expressed a greater confidence in the effects observed around the mean $PM_{2.5}$ concentrations in the long-term epidemiological studies; this does not necessarily imply a concentration threshold below which there are no effects. As such, these analyses are designed to transparently depict the additional uncertainty associated with $PM_{2.5}$-attributable risks estimated at lower concentrations. This cutpoint does not indicate a lower bound on the size of the forgone estimated ancillary health co-benefits. The cutpoint does not imply that forgone co-benefits are zero below the cutpoint; the forgone co-benefits could be greater or smaller than those estimated using a no-threshold concentration-response parameter.

Table 14. Summary of social costs and benefits (millions of 2016$) of the subcategorization in 2023 assuming that mortality risk $PM_{2.5}$-related benefits fall to zero below the lowest measured level of each long-term $PM_{2.5}$ mortality study, policy case compared to the primary baseline[a]

Description	Estimate (3% Discount Rate)	Estimate (7% Discount Rate)
Social benefits		
Cost reductions[b]	$33	$41
Social costs		
Forgone targeted benefits[c]	C	C
Forgone ancillary co-benefits above LML	$150 to $200	$140 to $180
Social net benefits		
Cost reductions minus forgone targeted benefits and forgone co-benefits	-$120 to -$160 - C	-$96 to -$140 – C

[a] Figures in this table are presented rounded to two significant figures. Totals may not sum due to independent rounding.
[b] The cost reductions presented in this section are estimates of the reductions in the social cost of the rule, as opposed to the reduction in private expenditures for affected EBCR-fired units presented in Section 2 of the memo, which was estimated to be $49 million in 2023. On an annualized basis, about 80 percent of the compliance cost reductions is associated with avoided capital expenditures.
[c] C is the sum of all unquantified forgone targeted acid gas HAP benefits (or costs in this table).

Table 15. Summary of social costs and benefits (millions of 2016$) of the subcategorization in 2023 when mortality risk $PM_{2.5}$ related benefits fall to zero below the $PM_{2.5}$ National Ambient Air Quality Standard, policy case compared to the primary baseline [a]

Description	Estimate (3% Discount Rate)	Estimate (7% Discount Rate)
Social benefits		
Cost reductions[b]	$33	$41
Social costs		
Forgone targeted benefits[c]	C	C
Forgone ancillary co-benefits above NAAQS	$4.0 to $7.2	$4.0 to $7.1
Social net benefits		
Cost reductions minus forgone targeted benefits and forgone co-benefits	$26 to $29 - C	$34 to $37 – C

[a] Figures in this table are presented rounded to two significant figures. Totals may not sum due to independent rounding.
[b] The cost reductions presented in this section are estimates of the reductions in the social cost of the rule, as opposed to the reduction in private expenditures for affected EBCR-fired units presented in Section 2 of the memo, which was estimated to be $70 million in 2023. On an annualized basis, about 80 percent of the compliance cost reductions is associated with avoided capital expenditures.
[c] C is the sum of all unquantified forgone targeted acid gas HAP benefits (or costs in this table).

When considering whether a regulatory action is a potential welfare improvement (*i.e.*, potential Pareto improvement), it is necessary to consider all impacts of the action. The presentation of forgone $PM_{2.5}$ ancillary co-benefits summarized above are part of one scenario of impacts arising out of the requirements of this final rule, which establishes a HAP source subcategory for coal-refuse facilities burning EBCR. The presentation is intended to capture the full incremental impacts arising out of this subcategorization when the baseline assumes full 2012 MATS compliance and specific control technologies.

The analysis does not account for how interaction with NAAQS compliance would affect the benefits (and costs) of the possible policy scenarios, which introduce uncertainty in the benefit (and costs) estimates. The EPA refers to the ancillary health benefits derived from reductions in emissions other than the listed HAP as “co-benefits” as they are not the targeted pollutant; the EPA may give reduction of targeted pollutants greater relative weight. The primary means by which the Clean Air Act regulates particulate matter is the PM NAAQS. The facilities impacted by this final rule will need to comply with any control obligations adopted by states to meet current and future PM NAAQS regulations.

6. Uncertainty

The analysis presented in this memo is subject to uncertainty. This section of the memo discusses key uncertainties that potentially affect the quantitative estimates of compliance cost reductions and forgone benefits that may result from this subcategorization.

a. Uncertainty in compliance cost reductions estimates

Two major sources of uncertainty in our compliance cost reductions estimates pertain to emissions control technology and operational assumptions. On the control side, there is uncertainty over both the technical feasibility of installing acid gas emissions controls at the EBCR-fired EGUs in the subcategory, as well as whether our compliance cost reductions estimates are too large or too small. In addition, there is uncertainty regarding the extent to which these units may continue to operate in the future, regardless of the MATS emissions limit the affected units may be subject to (either the 2012 MATS emissions limit, or the emissions limit established by the subcategorization).

i. Emissions control assumptions

To develop estimates of compliance cost reductions, this analysis assumes that EBCR-fired EGUs install dry scrubbers in the primary baseline. This is the same assumption made by the EPA in the 2012 MATS RIA. However, there is uncertainty over whether this control method represents a technically feasible approach for these units. As discussed in section III.A of the preamble for this final rule,[27] commenters claimed that post-combustion control strategies such as wet and dry scrubbers present installation difficulties given the layout of the facilities, local topography, and needs of the control system to interface with existing EGU equipment. To the extent that it is not technically feasible to install dry scrubbers, units may be forced to retire or seek alternate compliance options, such as polishing scrubbers (for which the EPA does not have sufficient cost information) or fuel-switching (which would reduce the amount of EBCR consumed and would thus interfere with the intended purpose of these EGUs).

The uncertainty regarding a compliance approach results in uncertainty regarding compliance cost reductions. For example, commenters note that one plant in the subcategory would be able to install a polishing scrubber ("enhanced all dry scrubber") for $4.97/MWhn, which is approximately 74 percent lower than the annual cost assumed for installation and operation of a traditional dry scrubber in this analysis.[28] If other plants in this subcategory are able to install

[27] See *National Emission Standards for Hazardous Air Pollutants: Coal- and Oil-Fired Electric Utility Steam Generating Units – Subcategory of Certain Existing Electric Utility Steam Generating Units Firing Eastern Bituminous Coal Refuse for Emissions of Acid Gas Hazardous Air Pollutants* in Docket ID No. EPA-HQ-OAR-2018-0794.
[28] Source: Comment from American Bituminous Power Partners, L.P. (Grant Town Power Plant), Docket ID No. EPA-HQ-OAR-2018-0794-1260.

these controls at a similar cost, it suggests that compliance cost reductions may be overestimated. On the other hand, and related to the technical feasibility point raised above, these units may possess site-specific characteristics that would significantly increase the cost of installing dry scrubbers beyond our estimates. For example, we modeled the plants with multiple EGUs as if they installed one system to cover emissions from both units. However, this assumption may cause us to underestimate the cost of a dry scrubber retrofit by improperly assuming economies of scale, particularly in the case of the Scrubgrass plant where each EGU currently emits through a separate stack.

ii. Operational assumptions

In addition to uncertainty regarding pollution control approaches, there is also uncertainty regarding the continued operation of EBCR-fired EGUs, under both the 2012 MATS emissions limits and the revised limits. In this memo, we assume in the primary baseline and policy scenario that each of the affected EBCR-fired EGUs continue to operate in 2023 as they did in 2016.[29] While this is an appropriate assumption for this analysis, note that the PJM market in which these units operate has become increasingly competitive in recent years.[30] Furthermore, this assumption implies that the demand for remediation and the location of coal refuse piles is sufficiently high and proximate enough to these EBCR-fired units to warrant their continued operation. Since we assume operation at 2016 levels in this analysis, the retirement of any of these EGUs in the baseline that would not be reversed with the subcategory would imply generally overestimated compliance cost reductions. An exception would be if the unit would retire in the baseline and policy scenario in less than 15 years, which is the assumed amortization period for the dry scrubber, yet still operate for a period with a dry scrubber. In this situation, the annual compliance cost reduction would be an underestimate.

b. Uncertainty in estimated forgone benefits

This analysis includes many data sources as inputs that are each subject to uncertainty. Input parameters include projected emission inventories, air quality data from models (with their associated parameters and inputs), population data, population estimates, health effect estimates from epidemiology studies, economic data for monetizing co-benefits, and assumptions regarding the future state of the world (*i.e.*, regulations, technology, and human behavior). When compounded, even small uncertainties can greatly influence the size of the total quantified benefits. In summary, the estimated health benefits from changes in $PM_{2.5}$ concentrations are subject to uncertainties related to: (1) the projected 2023 $PM_{2.5}$ concentrations; and, (2) the relationship between air quality changes and health outcomes; (3) the economic value of avoided air pollution-attributable premature deaths; (4) scaling health co-benefits by 65% to account for changes in the population of affected EBCR-fired units.

For the first uncertainty, we acknowledge that all models have some level of inherent uncertainty in their formulation and inputs. However, the base-year 2011 model outputs have been evaluated

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[29] Another source of uncertainty is that there may be a transition period in the baseline while the EBCR-fired EGUs adopt abatement technologies to achieve the current standard. During the transition period costs and benefits may differ than those estimated for 2023.

[30] See Docket ID No. EPA-HQ-OAR-2018-0794-1154.

elsewhere against ambient measurements (U.S. EPA 2017b; 2019b) and have been shown to adequately reproduce spatially and temporally varying $PM_{2.5}$ concentrations. Another limitation of the air quality modeling approach is the treatment of air quality changes from the tagged sources as linear and additive; this is consistent with past practices and is expected to reasonably represent the differences between the baseline and policy case.

We address the second uncertainty in part by quantifying benefits using two alternative adult mortality concentration-response relationships (*e.g.*, Krewski et al. (2009) and Lepeule et al. (2012)). The $PM_{2.5}$ concentration-response models assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality because the scientific evidence is not yet sufficient to allow differentiation of effect estimates by particle type. Furthermore, as discussed above, there is greater uncertainty in the effects of exposure at low $PM_{2.5}$ levels. Our estimate of the total monetized co-benefits is based on the EPA's interpretation of the best available scientific literature and methods and supported by the SAB-HES and the National Academies of Science (National Research Council 2002). Below are key assumptions underlying the estimates for $PM_{2.5}$-related premature mortality.

As noted above, we assume that the health impact function for fine particles is log-linear without a threshold. Thus, the estimates include forgone health co-benefits from reducing fine particles in areas with different concentrations of $PM_{2.5}$, including both areas that do not meet the fine particle standard and those areas that are in attainment and reflect the full distribution of $PM_{2.5}$ air quality simulated above.

We assume that there is a multi-year "cessation" lag between changes in PM exposures and the total realization of changes in health effects. Although the structure of the lag is uncertain, the EPA follows the advice of the SAB-HES to use a segmented lag structure that assumes 30 percent of premature deaths are reduced in the first year, 50 percent over years 2 to 5, and 20 percent over the years 6 to 20 after the reduction in $PM_{2.5}$ (U.S. EPA-SAB 2004). Changes in the cessation lag assumptions do not change the total number of estimated deaths but rather the timing of those deaths.

In general, we are more confident in the magnitude of the risks we estimate from simulated $PM_{2.5}$ concentrations that coincide with the bulk of the observed PM concentrations in the epidemiological studies that are used to estimate the benefits. Likewise, we are less confident in the risk we estimate from simulated $PM_{2.5}$ concentrations that fall below the bulk of the observed data in these studies. There are uncertainties inherent in identifying any particular point at which our confidence in reported associations decreases appreciably, and the scientific evidence provides no clear dividing line. This relationship between the air quality data and our confidence in the estimated risk is represented below in Figure 1.

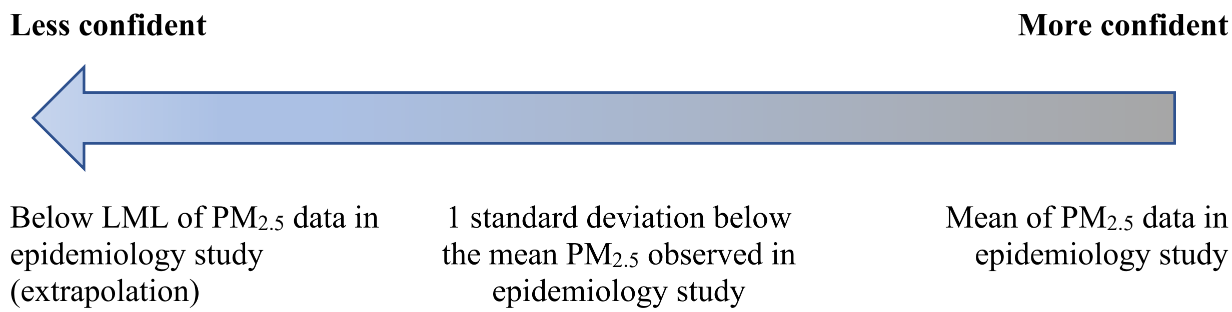

Figure 1. Stylized Relationship between the $PM_{2.5}$ Concentrations Considered in Epidemiology Studies and our Confidence in the Estimated PM-related Premature Deaths

In this analysis, we build upon the concentration benchmark approach (also referred to as the Lowest Measured Level analysis) that has been featured in recent RIAs and EPA's *Policy Assessment for Particulate Matter* (U.S. EPA 2011) by reporting the estimated PM-related deaths according to alternative concentration cutpoints.

Concentration benchmark analyses allow readers to determine the portion of population exposed to annual mean $PM_{2.5}$ levels at or above different concentrations, which provides some insight into the level of uncertainty in the estimated $PM_{2.5}$ mortality benefits. The EPA does not view these concentration benchmarks as concentration thresholds below which we would not quantify health co-benefits of air quality improvements.[31] Rather, the forgone co-benefits estimates reported in this memo are the most appropriate estimates because they reflect the full range of air quality concentrations associated with the emission increases being evaluated in this final rule. The PM ISA concluded that the scientific evidence collectively is sufficient to conclude that there is a causal relationship between long-term $PM_{2.5}$ exposures and mortality and that overall the studies support the use of a no-threshold log-linear model to estimate mortality attributed to long-term $PM_{2.5}$ exposure (U.S. EPA 2009).

Our approach to valuing avoided premature deaths is subject to uncertainty. The value of avoided premature deaths account for 98 percent of ancillary monetized PM-related forgone co-benefits. The economics literature concerning the appropriate method for valuing reductions in premature mortality risk is still developing. The value for the projected reduction in the risk of premature mortality is the subject of continuing discussion within the economics and public policy analysis community. Following the advice of the SAB's Environmental Economics Advisory Committee (SAB-EEAC), the EPA currently uses the value of statistical life (VSL) approach in calculating estimates of mortality benefits, because we believe this calculation provides the most reasonable single estimate of an individual's willingness to trade off money for changes in the risk of death (U.S. EPA-SAB 2000). The VSL approach is a summary measure for the value of small changes in the risk of death experienced by many people.

[31] For a summary of the scientific review statements regarding the lack of a threshold in the $PM_{2.5}$-mortality relationship, see the TSD entitled *Summary of Expert Opinions on the Existence of a Threshold in the Concentration-Response Function for $PM_{2.5}$-related Mortality* (U.S. EPA 2010a).

The EPA continues work to update its guidance on valuing mortality risk reductions, and the Agency consulted several times with the SAB-EEAC on this issue. Until updated guidance is available, the Agency determined that a single, peer-reviewed estimate applied consistently, best reflects the SAB-EEAC advice it has received. Therefore, the EPA applies the VSL that was vetted and endorsed by the SAB in the *Guidelines for Preparing Economic Analyses* (U.S. EPA 2016) while the Agency continues its efforts to update its guidance on this issue. This approach calculates a mean value across VSL estimates derived from 26 labor market and contingent valuation studies published between 1974 and 1991. The mean VSL across these studies is $6.3 million (2000$).[32] We then adjust this VSL to account for the currency year and to account for income growth from 1990 to the analysis year. Specifically, the VSLs applied in this analysis in 2016$ after adjusting for income growth is $10.5 million for 2025.

The Agency is committed to using scientifically sound, appropriately reviewed evidence in valuing changes in the risk of premature death and continues to engage with the SAB to identify scientifically sound approaches to update its mortality risk valuation estimates. Most recently, the Agency proposed new meta-analytic approaches for updating its estimates (U.S. EPA 2010b), which were subsequently reviewed by the SAB-EEAC. The EPA is taking the SAB's formal recommendations under advisement (U.S. EPA 2017).

Finally, applying a 65 percent scaling ratio to the forgone co-benefits estimate from the ten-unit modeling to estimate the foregone co-benefits from controlling the six remaining units adds uncertainty. This method does not explicitly account for the fraction of the impacted emissions that come from West Virginia versus Pennsylvania. Despite this limitation, we believe it is a reasonable approximation for the purpose of this rule for several reasons.

First, the forgone SO_2 emissions reductions occur in a relatively localized region in Pennsylvania and West Virginia, and the resulting sulfate concentration changes are also most pronounced in this region. Given that sulfate is a secondary pollutant, concentration changes from SO_2 reductions from EGUs are more regional in nature than impacts of primary pollutants which are more closely concentrated around the location of the emissions source. The changed operating status of the two plants (again, one in Pennsylvania and one in West Virginia) does not substantially change the region over which forgone SO_2 emissions reductions and resulting sulfate impacts occur.

Second, as explained in more detail in Appendix A, the analysis already relies on the assumption that sulfate impacts from small groups of EGUs are linearly related to SO_2 emissions from those same plants. By scaling the impacts, we lose some specificity in the initial analysis that differentiated between plants in West Virginia and Pennsylvania but still retain the general regional footprint of impacts across West Virginia and Pennsylvania.

By looking at the expected sulfate impacts from all EGUs in Pennsylvania and West Virginia and the relative fraction of SO_2 emissions originating from these two plants, we estimate that baseline annual average $PM_{2.5}$ concentrations are likely to change by less than 0.1 $\mu g/m^3$. Hence, when reporting the estimated number of forgone avoided $PM_{2.5}$-attributable deaths (Table 10),

[32] In 1990$, this base VSL is $4.8 million.

we find it appropriate to scale the estimated distribution of $PM_{2.5}$ attributable deaths at each concentration cutpoint by 0.65. Please note, as mentioned above, the locations seeing the largest impacts all have $PM_{2.5}$ concentrations less than 12 $\mu g/m^3$ in the baseline scenario.

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Appendix A: air quality modeling data and methods

The air quality model simulations (*i.e.*, model runs) were performed using the Comprehensive Air Quality Model with Extensions (CAMx) (Ramboll Environ 2016). Our CAMx nationwide modeling domain (*i.e.*, the geographic area included in the modeling) covers all lower 48 states plus adjacent portions of Canada and Mexico using a horizontal grid resolution of 12 x 12 km shown in Figure 2.

Figure 2. Air Quality Modeling Domain

As mentioned in Section 3 of this memo, the impact of specific emissions sources on $PM_{2.5}$ in the 2023 modeled case were tracked using a tool called "source apportionment." In general, source apportionment modeling quantifies the air quality concentrations formed from individual, user-defined groups of emissions sources or "tags". These source tags are tracked through the transport, dispersion, chemical transformation, and deposition processes within the model to obtain hourly gridded[33] contributions from the emissions in each individual tag to hourly modeled concentrations of $PM_{2.5}$.[34] Thus, the source apportionment method provides an estimate of the effect of changes in emissions from each group of emissions sources (*i.e.*, each tag) to changes in $PM_{2.5}$ concentrations. Examples of the magnitude and spatial extent of tagged contributions for $PM_{2.5}$ sulfate from coal-fired EGUs in West Virginia and Pennsylvania are provided in Figures 3 through 6 for January and July, respectively. These figures show how both the magnitude and the spatial patterns of contributions can differ by season.

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[33] Hourly contribution information is provided for each grid cell to provide spatial patterns of the contributions from each tag.

[34] Note that the sum of the contributions in a model grid cell from each tag for a pollutant equals the total concentration of that pollutant in the grid cell.

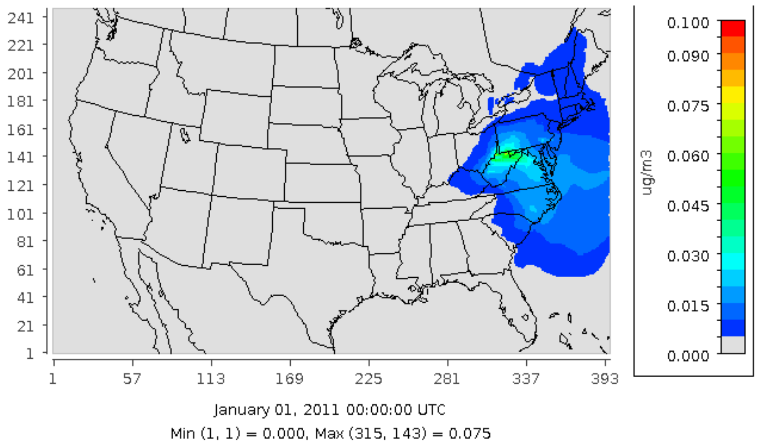

Figure 3. Map of West Virginia Coal EGU Tag Contributions to January Average Sulfate (µg/m³)

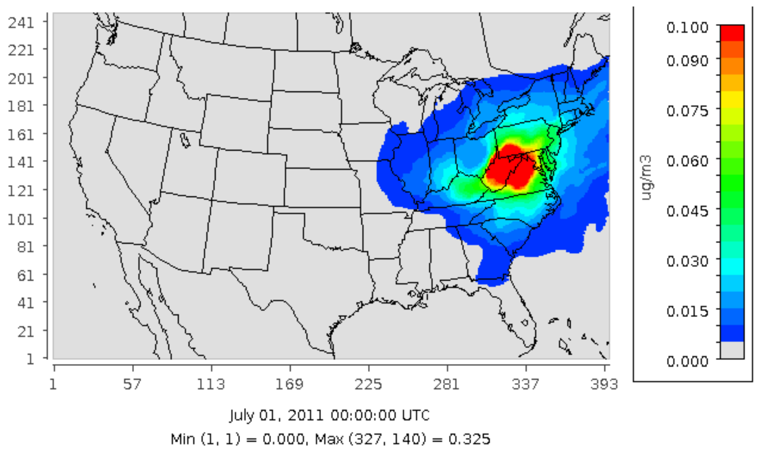

Figure 4. Map of West Virginia Coal EGU Tag Contributions to July Average Sulfate (µg/m³)

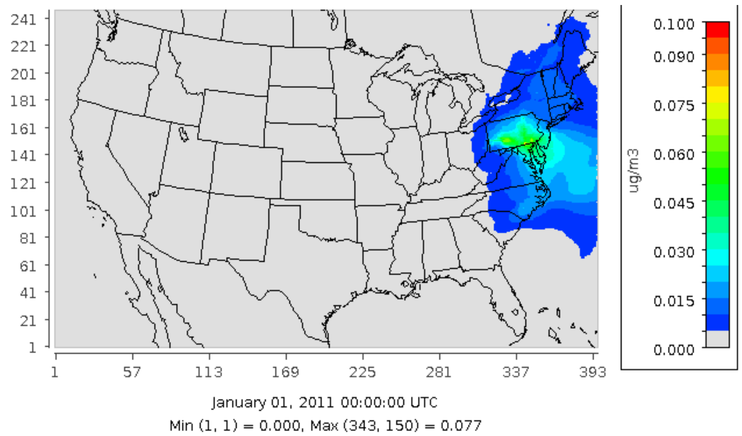

Figure 5. Map of Pennsylvania Coal EGU Tag Contributions to January Average Sulfate ($\mu g/m^3$)

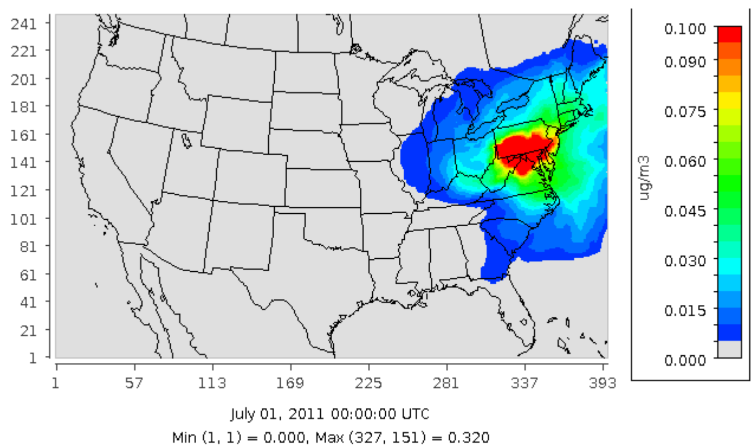

Figure 6. Map of Pennsylvania Coal EGU Tag Contributions to July Average Sulfate ($\mu g/m^3$)

For this analysis we applied outputs from source apportionment modeling for $PM_{2.5}$ using the 2023 modeled case (the policy case).[35] We used the Particulate Source Apportionment Technique (PSAT) tool in CAMx[36] to obtain the contributions from EGU emissions as well as other sources to $PM_{2.5}$ component species concentrations. The source apportionment modeling, which was already available from analysis performed to support U.S. EPA (2019b), was used to quantify the contributions from EGU emissions on a state-by-state or, in some cases, on a multi-state basis. For $PM_{2.5}$, we used modeled sulfate contributions from the 2023 EGU sector emissions of SO_2 for the entire year to inform the development of spatial fields of annual mean

[35] As the policy and baseline scenarios use actual 2016 emissions from all non-ECBR units that are expected to be operating, these scenarios do not reflect any implementation of the ACE rule.
[36] The Particulate Source Apportionment Technique (PSAT) tool is described in Ramboll Environ (2016).

$PM_{2.5}$. For each state, or multi-state group, we separately tagged EGU emissions depending on whether the emissions were from coal-fired units or non-coal units.[37] The ten ECBR-fired units in the original analysis were included in 4 separate tags in the available modeling: West Virginia coal EGUs, West Virginia non-coal EGUs, Pennsylvania coal EGUs, and Pennsylvania non-coal EGUs. As shown in Table 16, the ten ECBR-fired units accounted for almost all the SO_2 emissions in the West-Virginia "non-coal" EGU tag, and smaller fractions of SO_2 in the other three tags.

Table 16. Emissions information for source apportionment tags containing affected sources

Tag Name	Affected units included in this tag	Total SO_2 Emissions in Tag (thousand tons/yr in 2023 modeled case)	Fraction of SO_2 emissions in tag from affected ECBR sources
PA Non-coal EGU	Colver Power Project; Ebensburg Power Company; Cambria Cogen	19.6	0.297*
PA Coal EGU	Scrubgrass Generating Plant	66.5	0.017
WV Non-coal EGU	Grant Town Power Plant	2.4	0.999
WV Coal EGU	Morgantown Energy Facility	48.1	0.022

* The Cambria Cogen ECBR-fired units that have changed operating status account for approximately 13% of the Pennsylvania non-coal SO_2 emissions.

The source apportionment $PM_{2.5}$ contributions represent the spatial and temporal distribution of the emissions from each source tag as they occur in the 2023 modeled case. Thus, the contribution modeling results do not allow us to represent any changes to any "within tag" spatial distributions. For example, the affected EBCR-fired units in this rule only make up a portion of the total SO_2 emissions in the West Virginia Coal EGU tag. Since this method scales the entire tag contributions up uniformly based on total tagged emissions changes, it does not account for any changes of spatial patterns that would result from changes in the relative magnitude of sources within a source tag in the scenarios investigated here.

In addition to tagging state-level coal-fired and non-coal EGU emissions we also tracked the $PM_{2.5}$ contributions from the following "domain-wide" tags (*i.e.*, tags that are not geographically grouped by state or multi-state area): emissions from all of those EGUs in the 2023 emissions inventory that were operating in the 2023, but are now expected to retire before 2030;[38] U.S. anthropogenic emissions from source sectors other than EGUs; international emissions that are located within the modeling domain; emissions from wildfires and prescribed fires; biogenic source emissions; and contributions from concentrations along the outer boundary of the modeling domain.

[37] In the source apportionment modeling conducted, non-coal fuels included emissions from natural gas, oil, biomass, municipal waste combustion and in some cases, coal refuse EGUs.

[38] Note that emissions associated with units in the two EGU retirements tags are not included in the state-level EGU tags (*i.e.*, there is no double-counting of emissions contributions). While announced EGU retirements to take place between 2023 and 2030 are not relevant for the analysis of the current rule they were relevant to the analysis described in U.S. EPA (2019b).

The following data were used as inputs to create the spatial fields of $PM_{2.5}$ concentrations for the baseline and policy scenarios:

(1) 2023 modeled annual EGU SO_2 emissions associated with the four PA and WV source apportionment tags shown in Table 16;

(2) Annual EGU emissions of SO_2 for the baseline and policy[39] scenarios that correspond to each of the 2023 EGU tags shown in Table 16;

(3) Daily a) 2011 and b) 2023 modeling-based concentrations of 24-hour average $PM_{2.5}$ component species;

(4) 2023 daily contributions to 24-hour average $PM_{2.5}$ component species from each of the various source tags; and

(5) Base period (2011) "fused surfaces" of measured and modeled air quality[40] representing quarterly average $PM_{2.5}$ component species concentrations. These "fused surfaces" use the ambient data to adjust modeled fields to match observed data at locations of monitoring sites. Details on the methods for creating fused surfaces are provided in U.S. EPA (2019b, chap. 8).

Next, we identify the general process for developing the spatial fields for $PM_{2.5}$. First, we describe methods applied to create $PM_{2.5}$ surfaces associated with the policy scenario. This process requires fewer steps because the emissions scenario was directly modeled. Then we will describe methods applied to create $PM_{2.5}$ surfaces associated with the baseline scenario. This scenario requires additional steps because it relies on adjustments using the source apportionment contributions and associated emissions to estimate air quality changes compared to the modeled policy scenario.

The steps to create $PM_{2.5}$ surfaces associated with the policy scenario are as follows:

(1) We start with gridded daily concentrations for each $PM_{2.5}$ component species for the directly modeled 2023 modeled policy scenario (item 3b in list immediately above). For each $PM_{2.5}$ component species, we average the daily concentrations up to 3-month averages for each quarter of the year.

(2) The quarterly average $PM_{2.5}$ component species concentrations from step (1)[41] are divided by the corresponding quarterly average species concentrations from the 2011 CAMx model run (item 3a in the list above). This step provides a Relative Response Factor (*i.e.*, RRF) between 2011 and the policy scenario for each species in each model grid cell.

(3) The species-specific quarterly RRFs from step (2) are then multiplied by the corresponding species-specific quarterly average concentrations from the base period

[39] Policy scenario emissions are the same as emissions used for the modeled 2023 case.
[40] In this analysis, a "fused surface" represents a spatial field of concentrations of a particular pollutant that was derived by applying the Enhanced Voronoi Neighbor Averaging with adjustment using modeled and measured air quality data (*i.e.*, eVNA) technique (Ding et al. 2016).
[41] Ammonium concentrations are calculated assuming that the degree of neutralization of sulfate ions remains at 2011 levels (see Chapter 8 of U.S. EPA (2019b) for details).

(2011) fused surfaces (item 5 in the list above) to produce quarterly average species concentrations for the policy scenario.

(4) The policy scenario quarterly average species concentrations from step (3) are summed over the species to produce total $PM_{2.5}$ concentrations for each quarter.

(5) Total $PM_{2.5}$ concentrations for the four quarters of the year are averaged to produce the spatial field of annual average $PM_{2.5}$ concentrations for the policy scenario that are input to BenMAP-CE.

The steps to create $PM_{2.5}$ surfaces associated with the baseline scenario are as follows:

(1) We use the EGU annual SO_2 emissions for the baseline and the corresponding 2023 policy scenario to calculate the ratio of baseline emissions to 2023 policy scenario emissions for each for the four affected EGU tags (*i.e.*, an SO_2 emissions scaling ratio for each tag shown in Table 17).

(2) The tag-specific SO_2 emissions scaling ratios from step (1) are multiplied by the corresponding 365 daily 24-hour average gridded $PM_{2.5}$ sulfate contributions from the contribution modeling. This step results in 365 gridded surfaces of adjusted daily $PM_{2.5}$ sulfate contributions for four affected EGU tags to reflect the emissions in the baseline.

(3) The gridded surfaces of adjusted sulfate contributions for the four affected EGU tags from step (2) are added to unadjusted sulfate contributions from all other tags to produce a daily sulfate total. Steps 1-3 can be described using Equation 1:

$$Sulf_{g,d} = C_{g,d,BC} + C_{g,d,int} + C_{g,d,bio} + C_{g,d,fires} + C_{g,d,USanthro} + C_{g,d,EGU\_other} + \sum_{t=1}^{T} C_{g,d,t} S_t \qquad \text{(Eq-1)}$$

where:

- $Sulf_{g,d}$ is the estimated concentration for sulfate at grid-cell g on day d;
- $C_{g,d,BC}$ is the sulfate contribution from the modeled boundary inflow;
- $C_{g,d,int}$ is the sulfate contribution from international emissions within the model domain;
- $C_{g,d,bio}$ is the sulfate contribution from biogenic emissions;
- $C_{g,d,fires}$ is the sulfate contribution from fires;
- $C_{g,d,USanthro}$ is the sulfate contribution from U.S. anthropogenic sources other than EGUs;
- C_{g,d,EGU_other} is the sulfate contribution from all EGU tags other than those shown in Table 16;
- $C_{g,d,t}$ is the sulfate contribution from EGU emissions from tag t; and

- S_t is the sulfate scaling ratio for tag t.

(4) The gridded surfaces of daily total sulfate from step (3) are then combined with the species concentrations of all other $PM_{2.5}$ species from the 2023 model simulation. Note that we do not estimate changes in any $PM_{2.5}$ components other than sulfate as a result of SO_2 emissions changes using this method.

(5) For each $PM_{2.5}$ component species, we average the daily concentrations from step (4) for each quarter of the year to create quarterly-average gridded surfaces.

(6) The gridded surfaces of quarterly average $PM_{2.5}$ component species concentrations from step (5)[42] are divided by the corresponding gridded surfaces of quarterly average species concentrations from the 2011 CAMx model run. This step provides a Relative Response Factor (*i.e.*, RRF) between 2011 and the baseline for each species in each model grid cell.

(7) The gridded species-specific quarterly RRFs from step (6) are then multiplied by the corresponding species-specific quarterly average concentrations from the base period (2011) fused surfaces to produce quarterly average species concentrations for the baseline.

(8) The baseline quarterly average species concentrations from step (7) are summed over the species to produce gridded surfaces of total $PM_{2.5}$ concentrations for each quarter.

(9) Total $PM_{2.5}$ concentrations for the four quarters of the year are averaged to produce the spatial field of annual average $PM_{2.5}$ concentrations for the baseline that are input to BenMAP-CE.

Table 17. Baseline scenario sulfate scaling ratios for tags containing ten ECBR-fired sources operating in the initial analysis

PA Non-coal tag sulfate scaling ratio ($S_{s,t,i}$)	PA Coal tag sulfate scaling ratio ($S_{s,t,i}$)	WV Non-coal tag sulfate scaling ratio ($S_{s,t,i}$)	WV Coal tag sulfate scaling ratio ($S_{s,t,i}$)
0.8148	0.9941	0.4123	0.9921

One thing to note with respect to emissions from the affected tags (Table 17), is that the two PA tags and the WV Coal tag emit substantially more SO_2 than the WV non-coal tag. However, the fraction of the WV non-coal tag emissions coming from EBCR-fired units is much higher. Consequently, the scaling ratios show in Table 17 are closer to 1 for the two PA tags and the WV non-coal tag than they are for the WV non-coal tag. This accounts for the expected impact on EBCR-fired EGU emissions from this policy as well as the relative importance of those emissions to the tag.

One limitation of the scaling methodology described in steps 1-3 for creating $PM_{2.5}$ surfaces associated with the baseline or policy scenarios described above is that it treats air quality

[42] Ammonium concentrations are calculated assuming that the degree of neutralization of sulfate ions remains at 2011 levels (see Chapter 8 of U.S. EPA (2019b) for details).

changes from the tagged sources as linear and additive. It therefore does not account for nonlinear atmospheric chemistry and does not account for interactions between emissions of different pollutants and between emissions from different tagged sources. This is consistent with how air quality estimations have been treated in past regulatory analyses (U.S. EPA 2012b; 2019b). We note that emissions changes between scenarios are relatively small compared to 2023 totals from all sources. Previous studies have shown that air pollutant concentrations generally respond linearly to small emissions changes of up to 30 percent (Dunker et al. 2002; Napelenok et al. 2006; Koo, Dunker, and Yarwood 2007; Cohan and Napelenok 2011) and that linear scaling from source apportionment can do a reasonable job of representing impacts of 100 percent of emissions from individual sources (Baker and Kelly 2014). Therefore, while simplistic, it is reasonable to expect that the differences between the baseline and policy scenarios can be adequately represented using this methodology.

Below we present the model-predicted changes in annual mean $PM_{2.5}$ concentrations between the baseline and the policy case (Figure 7) using all ten ECBR-fired units. The map of changes displays the change in annual average $PM_{2.5}$ calculated as the policy case minus the baseline. The largest changes in $PM_{2.5}$ concentrations are estimated to occur in Pennsylvania and West Virginia with portions of these states expected to experience $PM_{2.5}$ concentrations 0.01- 0.1 µg/m^3 higher in the policy case than in the baseline. The spatial patterns shown in the figure are a result of (1) of the spatial distribution of sources and emissions for the four source apportionment tags which contain EBCR-fired units and (2) of the physical or chemical processing that the model simulates in the atmosphere. The spatial fields used to create this map serves as an input to the benefits analysis.

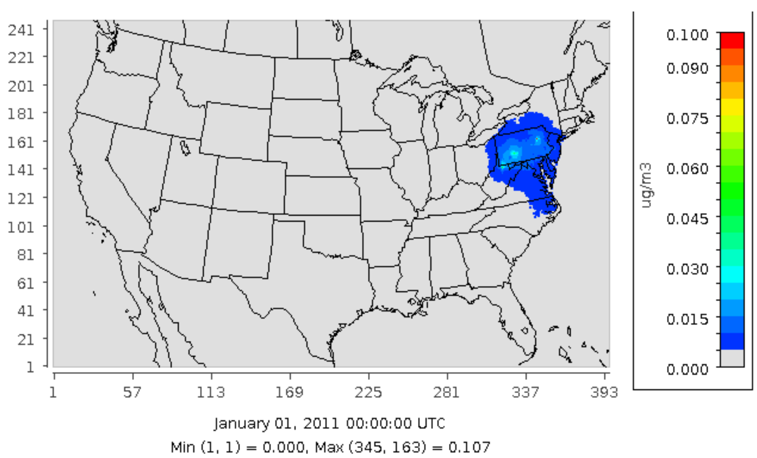

Figure 7. Change in annual mean $PM_{2.5}$ (ug/m^3): policy case – baseline, estimated to be attributable to the subcategorization

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