Appendix G

Comparison of Previous and Current Modeling of Particulate Matter at Two Cement Plants

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In support of cement kiln dust (CKD) regulatory activities, EPA has, at two different times, conducted modeling to predict particulate matter (PM) concentrations at cement manufacturing plants to assess the risks posed by fine particulates. Both approaches used fate and transport modeling to quantitatively estimate the potential effects associated with current CKD management practices on-site at cement plants, but each followed different methods, used different parameters, and yielded different results. The most recent round of modeling, completed initially in October 1996, predicted PM concentrations for two facilities: Rinker (Miami, FL) and Lafarge (Fredonia, KS). The previous analysis, reported in August, 1994,[1] also modeled the PM concentrations for these two facilities plus three others: Ash Grove (Chanute, KS), Southdown (Lyons, CO), and Giant (Harleyville, SC). Since Rinker and Lafarge have been modeled twice, the purpose of this appendix is to identify and explain the differences between the two sets of results and the different modeling approaches.

Results from the Two Modeling Exercises

The new modeling exercise provided a more complete profile of PM concentrations at the two facilities than the previous exercise. The new study predicted concentrations for two sizes of PM (i.e., PM_{10} and $PM_{2.5}$) over two different averaging periods (i.e., 24-hour and annual). The previous effort only predicted PM_{10} concentrations for the annual averaging period. The previous effort did, however, generate both a "best estimate" and an "upper bound" exposure concentration for each receptor point at each facility, based on best estimate and more conservative characterizations of the management and environmental parameters contributing most to the model results. Only "best estimates" were generated by the new study. Accordingly, the results of the new study and the previous study overlap only for the "best estimate" annual PM_{10} concentrations. These two sets of results are presented in Exhibit G-1.

Before drawing conclusions about which results are higher or lower, it is necessary to account for differences in (among other factors discussed in the remainder of this appendix) the modeled distance between the emission source and the exposure point. The previous study estimated concentrations at the nearest property boundary and the nearest residence. The new study, in contrast, estimated concentrations at points defined by a fixed grid around the facilities, which do not necessarily correspond to the points modeled previously. For the purpose of this comparison, EPA selected results from the new

[1] *Technical Background Document for the Notice of Data Availability on Cement Kiln Dust: Human Health and Environmental Risk Assessment in Support of the Regulatory Determination on Cement Kiln Dust.* EPA Office of Solid Waste, August 31, 1994.

modeling that correspond most closely to the distances modeled previously, but there are differences which contribute to the varied results.

Exhibit G-1
Comparison of Model Results

	Annual PM_{10} Concentration ($\mu g/m^3$)	
Facility:	**Rinker (Miami, FL)**	**Lafarge (Fredonia, KS)**
Current Model	0.4-0.5	40-43
Previous Model	1-6	40-50

In the case of Rinker, the distances modeled are quite similar: 701 and 1,069 meters in the old analysis compared to 605 and 927 meters in the new analysis. Based on these similarities, it is reasonable to compare the results directly. This leads to the overall conclusion that the new modeling at Rinker predicts lower concentrations (by about an order of magnitude) than the old modeling.

In the case of Lafarge, the distances modeled in the new study (880 and 1,145 meters) are approximately twice as far as those modeled in the old study (457 and 488 meters). The predicted concentrations from the two different exercises, however, are almost equivalent. Because concentrations are expected to be higher at points closer to a ground-level source, such as a CKD pile, it can be inferred that the new results would be higher than 40-43 g/m^3 – and thus higher than the concentrations predicted previously – if the new model were re-run for the closer distance modeled previously. This leads to the overall conclusion that the new modeling at Lafarge predicts higher concentrations than the old modeling.

Discussion of Main Factors and Direction of Impact

To explore why the new results for Rinker are lower and the new results for Lafarge are higher, the main differences between the two modeling exercises are discussed below. While the precise magnitude of effect that each of these modeling differences has on the results is beyond the scope of this brief review, a general estimation of the direction of impact can be predicted in most cases (i.e., would the difference tend to make the new model's results higher or lower than the previous model's results). First, differences in the basic models used are discussed. Next, the specific modeling parameters, how they differ between the two approaches, and the direction of impact of each factor are presented.

Models

The two studies used different models. These differences are described below first for the emissions models (which estimate the amount of PM that escapes from a source) and then for the dispersion models (which estimate the change in air concentration from source to receptor).

Emissions Models

The new study estimated emissions using emission factors and equations found in AP-42.[2] The methods presented in AP-42 for estimating fugitive dust emissions are principally compiled from a 1988 reference.[3] This is the approach that EPA's Office of Air Quality Planning and Standards recommends today. It is the most up-to-date and best approach for estimating emissions, short of substantially more complicated modeling or new field studies.

In the previous effort, MMSOILS was used to estimate releases to the atmosphere from CKD piles. MMSOILS is a screening-level multimedia model developed by EPA's Office of Research and Development (ORD) to simulate the release of hazardous constituents from a wide variety of waste management units and their subsequent transport through different environmental pathways.[4] The air emissions component of MMSOILS consists of equations presented in a 1985 rapid assessment methodology for estimating potential atmospheric contamination and resulting inhalation exposure of people living near abandoned hazardous waste sites.[5] The particular equation used in the previous CKD modeling exercise, which was developed from field measurements of highly erodible soils, estimates PM_{10} emissions as a function of certain field and climatic factors, including the fraction of the surface covered with vegetation and the wind speed. This equation is older and different than the equations recommended in AP-42.

EPA has not systematically compared these two approaches for estimating emissions or compared results from the two approaches under a common set of assumptions. Therefore, the Agency recognizes the use of different emission models as a source of variation between the two sets of results, but the direction and magnitude of the impact is presently unknown.

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[2] *Compilation of Air Pollution Emission Factors, AP-42,* Fifth Edition. EPA, Research Triangle Park, North Carolina, January 1995.

[3] *Control of Open Fugitive Dust Sources*, Final Report. Prepared by C. Cowherd *et al.* of Midwest Research Institute for EPA Office of Air Quality Planning and Standards, EPA-450/3-88-008, September 1988.

[4] *MMSOILS: Multimedia Contaminant Fate, Transport, and Exposure Model, Documentation and User's Manual.* EPA Office of Research and Development, September 1992 (updated in April 1993).

[5] *Rapid Assessment of Exposure to Particulate Emissions from Surface Contamination Sites.* EPA Office of Health and Environmental Assessment, EPA/600/8-85/002, February 1985.

Dispersion Models

The new study used the Industrial Source Complex 3-Short Term (ISC3ST) dispersion model. This model is recommended in the EPA Guidelines on Air Quality Models for dispersion modeling of complex industrial source facilities. The ISC3ST model is a significant improvement over the model used in the previous study. For example, ISC3ST can discriminate between airborne particulate concentrations that are due to emissions from the CKD pile versus those from the CKD handling train. Also, the ISC3ST model can incorporate hourly wind speeds into calculations of dispersion, uses as input the full array of meteorological data, and handles terrain effects (e.g., the model accounts for the difference in elevation between the source and various receptor points).

The previous study differed in terms of both the model and inputs used. The model itself, a component of MMSOILS, was a sector-averaged form of a Gaussian plume model. It did not account for particulate emissions from the handling train, terrain effects, and other complexities considered in the more recent study. Similarly, rather than run the model using a complete stability array of meteorological data, a simplified set of conservative atmospheric transport parameters was assumed based on recommendations from ORD. These assumptions included a stability class of E, a wind speed of 3 meters/second, and a frequency of wind blowing in any one direction of 30 percent.

As for the emission models discussed above, the Agency has not systematically compared these two dispersion model approaches to know which one yields higher or lower results, and by what margin. However, the new study used a model and set of inputs that are believed to be more sophisticated and more accurate than the previous study.

Specific Model Factors

Emissions and dispersion factors used in the two analyses are discussed below. By "factor" EPA means inputs to the models, modeling approaches, or underlying assumptions of the models. The primary factors are listed and described below, along with predictions of the direction of influence of each factor on estimated PM concentrations.

Emissions Factors

Source of PM. The new modeling predicted PM emissions from both the CKD pile (through wind erosion and vehicle disturbance) and the handling train (including initial loading and unloading, interim storage, and transport to the CKD pile). The previous model was limited to emissions from the pile, but included both wind erosion from the pile and particulate emissions resulting from vehicular disturbance from trucks delivering CKD to the pile and from the associated spreading operations.

Direction: The new modeling would be expected to estimate greater PM emissions because it considers the handling train in addition to the other sources evaluated previously (wind erosion and vehicle disturbance at the pile). In particular, fugitive dust kicked up from roads used to transport CKD to the disposal pile was found in the new study to add significantly to the PM emissions. At Lafarge, fugitive dust from the road was

estimated to contribute 88 percent of the emissions, whereas the other CKD handling stages contributed 2 percent and wind erosion from the CKD pile contributed 10 percent.

Covering the pile. For Lafarge, the new study accounted for the fact that only 8,600 m^2 (7.5 percent) of the 114,000 m^2 pile is uncovered (the rest has been covered with soil and seeded). The previous study modeled the entire pile as being exposed and contributing to PM emissions. (Both analyses treated the pile at Rinker as completely uncovered.)

Direction: The new estimate of PM_{10} emissions at Lafarge would be more realistic but lower than the previous estimate. (Not applicable for Rinker.)

Active pile surface. The new study estimated continual emissions from only the active part of the CKD disposal pile, where fresh or disturbed CKD without a surface crust is highly susceptible to wind erosion. Emissions from inactive areas of the pile were simulated until erodible particles on the surface were blown off and a surface crust forms and prevents further emissions. In contrast, the previous study treated the entire surface of the pile as an unlimited supply of erodible particles.

Direction: All other factors being equal, this aspect of the new study results in lower emission estimates than the previous study.

Dust suppression. For the new study, EPA identified the type of dust suppression used at each facility and reduced the predicted PM emissions according to the estimated effectiveness of the suppression technique. In particular, at Lafarge EPA assigned a 50 percent emissions reduction during CKD handling due to water addition (nodulization) at the plant prior to transport of CKD to the pile. At Rinker EPA did not identify any emissions controls, so no credit was given. The previous study did not account for any dust suppression.

Direction: Giving credit for emission controls at Lafarge in the new study would tend to reduce emissions estimates compared to the previous study. No such differences would be expected for Rinker since no emissions reduction credits were assigned in the new study or the previous study.

Daily load of CKD to the disposal pile. The new modeling computed the daily load by dividing the quantity of CKD wasted annually by the number of operating days in a year (for Rinker, 340 working days per year; for Lafarge, 339 working days per year). The previous study used the same approach but set the number of working days per year at 300 for all facilities.

Direction: This factor works in two ways. First, the larger number of working days used in the new study results in lower daily loads and, thus, lower daily emissions than the previous study. This difference, however, does not affect emissions over the course of an entire year, which is the relevant period for comparison of the results in Exhibit G-1. Second, more working days per year results in more frequent trips to the pile and more frequent disturbances of the pile surface. This second effect results in the new study estimating higher annual emissions than before.

Truck capacity. The initial new modeling for Rinker and Lafarge assumed a truck can carry 36 tons of CKD, which is the loaded (truck plus payload) weight of a typical dump truck used in hauling operations, according to truck brochures. (Note that this value

was refined in subsequent stages of the new modeling, as described in Section 3.2.5 of the main body of this report.) The previous study assumed a capacity of 80 tons of CKD per truckload.

Direction: The smaller truck capacity in the new study results in more trips occurring between the plant and the pile and more frequent disturbances of the pile surface, which result in higher emissions estimates than the previous modeling.

Particle size. In the new study, EPA used particle size data developed by the Portland Cement Association for CKD from three different kinds of kilns (long wet rotary, long dry rotary, and precalciner system).[6] The type of kiln actually used at Rinker and Lafarge was then used to select the most representative particle size distribution. In the previous study, engineering judgment was used to estimate a mean particle size, based on available data for CKD-like material (fine silt).

Direction: The new modeling presumably provides more accurate results than the previous modeling; however, it is unclear in which direction (i.e., higher or lower concentration) the results would be influenced.

Dispersion Factors

Meteorological parameters. As mentioned previously, the new modeling used the full set of stability array data from the meteorological station nearest each facility. A complete set of stability array data consists of a joint frequency distribution of twelve wind directions, five wind speed categories, and six Pasquill-Gifford stability classes, resulting in a matrix of 360 entries with unique frequencies of occurrence. Due to the nature of the previous screening analysis, a complete set of meteorological data was not used in the atmospheric dispersion modeling. Instead, a simplified set of conservative atmospheric transport parameters was used based on recommendations from EPA's ORD. Key components of the meteorological array are discussed and compared below.

- Wind direction: The new dispersion modeling used actual wind direction data from the nearest meteorological station. The previous modeling assigned a 30 percent probability that the wind will be in any given direction.

- Wind speed: Again, the new modeling used actual wind speed measurements from the nearest meteorological station. The previous modeling assumed a wind speed of 3 meters/second, which was recommended by EPA's ORD as a reasonably slow wind not likely to result in significant dispersion.

- Stability class: The new modeling used data from the closest meteorological station to determine the stability class (the range of possible classes is A-F). The previous modeling conservatively used the "slightly stable" stability class (class E).

In each of these cases, the site-specific data used in the new modeling result in a more accurate characterization of dispersion conditions than before. Because the assumed

[6] *Cement Kiln Dust Management Permeability*. By H. Todres *et al.* for the Portland Cement Association. PCA Research and Development Bulletin RD103T, 1992.

set of conditions used in the previous modeling was purposefully selected to be conservative (i.e., estimate low dispersion), the values used previously likely result in higher PM concentrations than the values used in the new modeling.

Terrain. The new modeling used site-specific terrain features as a model input. The previous modeling did not include any adjustments for terrain.

Direction: The effects of terrain can be complex; the direction of impact on the new modeling results is unknown.

Conclusion

Results from the new modeling exercise should be more accurate than the previous modeling results. The emissions and dispersion models used in the new study are more sophisticated, the CKD handling and disposal practices at each site were characterized more thoroughly, and site-specific meteorological and terrain data were used as model inputs instead of conservative default assumptions. All of these changes are improvements over the earlier approach.

On a general level, the two exercises yield the same conclusion. The annual PM_{10} concentrations predicted at the closest modeling points, in both exercises, do not exceed the corresponding National Ambient Air Quality Standard of 50 $\mu g/m^3$.

There are so many differences in approach in the two studies, with each likely to affect the results to a different degree and in a different direction, that it is difficult to precisely explain the varied results. Overall, one might expect the new PM concentrations to be lower than those estimated previously, using the simpler and generally more conservative approach. This is the case for the Rinker results. The new PM concentrations predicted at Lafarge, however, are higher than before, when differences in the modeling distance in the two studies are taken into account. This result for Lafarge would appear inconsistent, especially given the fact that some of the changes in the new study (covering the pile, active pile surface, dust suppression) would tend to drive the estimated concentrations down from before. The principal offsetting change, which would tend to increase predicted concentrations, is the added emissions from the CKD handling train considered in the new study. In particular, fugitive dust from the road used to haul CKD from the plant to the pile at Lafarge was estimated to be a significant source of emissions in the new study.

These levels are intended to assure that high end risk levels do not exceed 1E-05 or HQs of 1 for any constituent regardless of the agricultural practices employed. These values are estimated by assuming all agricultural practice parameters are high end and estimating the concentration required to reach the desired risk or HQ levels. The estimated values were then inserted into the deterministic risk analysis with all the agricultural practice parameters assumed to be high end. All exposure parameters are varied singly or doubly in the remainder of the deterministic analysis. The resulting risk and HQ values are compared to the target values and the constituent concentrations adjusted to determine the limiting values.

This process is straight forward for the metal constituents. However, for dioxins it is not appropriate to set a limiting value for for each congener independently. The risk for dioxin congeners are determined using the TEF methodology and although congener concentrations and risk are estimated individually, the risks from individual congeners are summed to produce single TEF risk value for all dioxins and furans. The limiting concentration of dioxin congeners is estimated using the congener showing the highest risk (1,2,3,4,7,8-hexchlorodibenzodioxin) to set the limiting value for the TEF risk in the same way it is done for metals. The ratio of this estimated concentration to the highest measured value for that congener was found to be 4.5. The factor of 4.5 was then applied to the 95th percentile of the measured concentration of all congeners. The resulting concentrations were then used as the limiting values in the risk analysis.

* congener w highest risk →