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Energy and Environmental Engineering Center

Reconciling Urban Fugitive Dust Emissions Inventory and Ambient Source Contribution Estimates: Summary of Current Knowledge and Needed Research

DRI Document No. 6110.4F

May, 2000

PREPARED BY

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Preface and Disclaimer

This document results from a workshop conducted with a selection of experts in Las Vegas, NV, on July 15-17, 1998. The challenge presented to these experts was to explain, to the extent possible with existing information, discrepancies between relative amounts of fugitive dust in PM_{10} and $PM_{2.5}$ emissions inventories and in source apportionment studies of ambient air samples. These experts formulated various hypotheses and supplied information from their previous research and experience that supported or refuted these hypotheses. They also identified areas where information is lacking and identified ways to fill those information gaps.

The contributions from these experts have been assembled, edited, and presented in this document by the authors. The conclusions drawn and recommendations made do not necessarily represent the views of every contributor or of the U.S. Environmental Protection Agency (EPA). The primary authors have attempted to faithfully integrate and explain the issues concerning reconciliation of emissions inventory and ambient source apportionment measurements. They have drawn primarily on their own experience in presenting examples and do not represent this document as a comprehensive review of any topic. A comprehensive bibliography has been assembled and included that could serve as a starting point for several in-depth reviews of the topics presented here.

On May 14, 1999, the U.S. Circuit Court of Appeals for the District of Columbia issued a ruling that remanded the ozone and particulate matter standards. The Court also called into question what actions, if any, may be taken to implement the new standards. The development of Statewide emissions inventories for ozone and particulate matter and their precursors is necessary to address regional issues, irrespective of the final determination on the actual National Ambient Air Quality Standards. Since these are criteria pollutants and are key components of regional haze, development of emissions inventories is still deemed to be appropriate including the statewide periodic emissions inventories for calendar year 1999. Thus, it is appropriate for the Regions and State/local agencies to: 1) discuss plans for developing emissions inventories for 1999; 2) identify appropriate future actions, programs and associated milestones regarding emissions inventories; and 3) identify specific issues that need resolution prior to making progress on emissions inventories.

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EXECUTIVE SUMMARY

Source apportionment studies show that geological material contributes an average of ~40% to more than 60% of PM_{10} (particles with aerodynamic diameters less than 10 µm) and ~5% to ~20% of $PM_{2.5}$ (particles with aerodynamic diameters less than 2.5 µm) in urban areas where National Ambient Air Quality Standards (NAAQS) have been or might be exceeded. Urban emissions inventories show dust emissions contributing ~70% to ~90% of primary PM_{10} and ~50% to ~80% of $PM_{2.5}$. Geological contributions to ambient measurements are often, but not always, overestimated by dispersion models that simulate contributions to receptor concentrations.

A workshop of experts was convened to identify differences between emission estimates and ambient geological contributions, estimate the magnitudes of these differences that could be caused for different reasons, evaluate the technical bases for currently applied emission estimation procedures and recommend activities that can be conducted immediately, in the short term (1 to 2 years), and over a longer term (2 to 5 years) to improve scientific understanding of and quantitative estimation of fugitive dust emissions.

One of the methods used to identify potential discrepancies was to compare emissions estimated for a single state, California, with national inventory estimates for that state. The methodology is essentially the same for these inventories, but the emission factors, activity databases, and control measures applied are substantially different. This results in California's urban dust emissions from unpaved roads, paved roads, and construction being only 60% of those estimated for the state in the national inventory. Much larger discrepancies were found for specific areas and source types.

Conclusions

- National inventories do not accurately estimate emissions from fugitive dust sources that affect ambient $PM_{2.5}$ and PM_{10} concentrations for a variety of reasons. Both negative and positive biases in emissions result, but the net effect is to overestimate emissions from fugitive dust sources relative to those from other sources in the inventory.

- Suspendable particles are not transportable particles. Available data shows that ~75% (ranging from ~60% to ~90%) of suspended PM_{10} remains within 1 to 2 m above ground level. These particles deposit to the surface or impact on nearby vertical structures within a few minutes after suspension. Although horizontal fluxes commonly used in empirically-derived emission factors represent the mass of dust particles suspended from a surface, they do not represent the mass entrained into the atmosphere and transported over distances of more than a few kilometers.

- Source and receptor models do not represent the same spatial and temporal scales and emissions inventories. Better integration with and interaction among emissions, meteorological, chemical/transport, and receptor models is needed.

This interaction should help to identify and reconcile deficiencies in all of the modeling components.

- State and local values for emission factors and activities yield lower emission estimates, at least for California. Several of these emission factors and activities should be used in national inventories.

- Emissions inventories treat fugitive dust emissions as continuous processes, whereas they are intermittent processes that depend on many meteorological and activity variables. This causes a positive bias in dust emission estimates.

- Few empirical tests are available for the $PM_{2.5}$ fraction of fugitive dust emission factors. The extent to which total suspended particles (TSP) and PM_{10} emission factors can be scaled to $PM_{2.5}$ is uncertain.

- Paved and unpaved road dust emission factors have been found to be of similar form and magnitude for many independent tests for TSP and PM_{10}. Silt loadings, silt fractions, and activity levels used in national inventories appear to overestimate emissions from these sources, at least with respect to California.

- Existing tests for emission factors are biased toward the highest emitters, yet they are applied to a population of activities that encompasses a wide range of emission magnitudes. Low to moderate emitters are poorly characterized. This is especially true for construction emission factors that characterize the earth moving portion of a project and not the other phases of a construction project.

Immediate Recommendations for National Emissions Inventory Improvement

- Review construction emission factors and compare to those used in California.

- Use measurements from previous fugitive dust emission tests to estimate horizontal dust fluxes above elevations of 2 m. Create separate estimates for emissions above these heights that can be used for different modeling and planning purposes. Urban- and regional-scale source or receptor modeling could use the >2 m above ground level horizontal fluxes. Preliminary estimates given in this report indicate that >2 m horizontal fluxes are ~25% of total horizontal fluxes.

- Conduct countywide comparisons between national inventory and independently compiled statewide inventories. Use the results of these comparisons to identify the availability of or need for local information that affect suspendable dust surface loadings and dust-generating activities.

Short-Term Recommendations for National Emissions Inventory Improvement

- Create and apply methods to estimate vertical flux, as well as horizontal flux, from fugitive dust sources and add vertical flux estimates to modeling inventories. Revise the current emission measurement methodology such that vertical fluxes and horizontal fluxes above selected elevations are reported in inventories.

- Conduct additional emission tests of unpaved roads, paved roads, construction, and other earth-moving emissions that apply a vertical and horizontal flux method and are specific to $PM_{2.5}$ and PM_{10} particle sizes. These tests should be planned to represent a variety of areas in the eastern and western United States.

- Conduct detailed studies of temporal variability for underlying activities (including control measure effectiveness) that create dust. Determine representative temporal profiles for diurnal, weekly, monthly, and annual emissions. Determine the extent to which reservoirs of dust are depleted and incorporate these into emission models.

- Create a modeling framework that integrates emissions, meteorological, chemical/transport, and receptor models. This framework would use the meteorological model to estimate meteorological effects such as wind speed and moisture that affect fugitive dust emission rates. Receptor models applied to ambient data would be used to estimate source contributions at receptors and to reconcile emissions with these contributions on an area-specific basis.

Long-Term Recommendations

- Modify existing air quality modeling software to better represent the vertical flux, deposition, and transport for several spatial scales. Develop, test, and apply more realistic middle- and neighborhood-scale mathematical models to represent dust concentrations at receptors near a variety of dust emitters.

- Develop a complete, GIS-based, emission model that can be applied to neighborhood, urban, and regional scales. This model would include and update commonly available activity databases such as land uses, roadways, and soil surveys. It would provide for the use of temporal profiles that allow for daily, weekly, monthly, and annual emission estimation. The model would also contain fugitive dust source profiles suitable for receptor model source apportionment and for creating speciated inventories that can be used for rollback modeling, propagate input uncertainties, project emissions under different development scenarios, and allow for alternative emission factors and activity databases to estimate the same emission rates.

- Compile dust characteristics, including surface loadings, chemical compositions (source profiles), and suspendable particle content (silt or other) for representative locations throughout the U.S. Include these in a documented database and apply

them to development of emission estimates. Priorities should be given to airsheds that operate $PM_{2.5}$ chemical speciation monitors.

- Calculate source contributions with receptor models applied to data from the national $PM_{2.5}$ speciation network. Reconcile these contributions with fugitive dust emission estimates from these areas and with background concentrations from nearby IMPROVE monitoring sites. Use the results of this reconciliation to further identify and ameliorate emission modeling deficiencies.

- Identify and characterize chemical or physical components in different fugitive dust sources that allow fugitive dust sub-types to be distinguished from each other in receptor samples. Apply these characterization methods to receptor samples, quantify contributions at representative distances from sub-type emitters, and use the results to improve fugitive dust emission estimation methods.

- Define and conduct experiments that increase understanding of vertical flux, deposition, and removal by surrounding barriers. Such experiments might include eddy-correlation micrometeorological measurements of positive and negative vertical fluxes, balances of airflow through and over tree stands downwind of dust emitters, and measures of particle rebound and filtration over short vegetation, such as grass.

- Develop and apply novel methods to estimate fugitive dust emissions using real-time and remote sensing methods. These methods would be used to verify emissions from the more established and commonly applied methods as well as to better understand the physical interactions between emissions and the atmosphere.

1. INTRODUCTION

1.1 Statement of Problem and Objectives

Source apportionment studies show that geological material contributes an average of ~40% to more than 60% of PM_{10} (particles with aerodynamic diameters less than 10 µm) and ~5% to ~20% of $PM_{2.5}$ (particles with aerodynamic diameters less than 2.5 µm) in urban areas where National Ambient Air Quality Standards (NAAQS) have been or might be exceeded. Urban emissions inventories show dust emissions contributing ~70% to ~90% of primary PM_{10} and ~50% to ~80% of $PM_{2.5}$. Geological contributions to ambient measurements are often, but not always, overestimated by dispersion models that simulate contributions to receptor concentrations. *(Note: See the preface and disclaimer page for information on recent court decisions involving the PM NAAQS.)*

This report summarizes and evaluates existing knowledge of urban fugitive dust emissions, their interaction with the atmosphere, and ambient dust concentrations. It intends to determine the most probable causes of these discrepancies and to estimate the degree to which urban dust emissions are overestimated relative to other source emissions. Emissions due to wind erosion of non-urban surfaces are not included in this assessment. Objectives are to:

- State and test hypotheses, based on existing information, for differences between emission estimates and ambient geological contributions.
- Estimate the magnitudes of differences between emissions inventories and fugitive dust contributions to ambient air that are consistent with hypotheses.
- Evaluate the technical bases for currently applied emission estimation procedures and their consistency with the hypotheses.
- Identify scientifically valid changes to urban fugitive dust emission methodology to support modeling and source assessment.
- Specify research projects for the short term (1 to 2 years) and long-term (2 to 5 years) that will improve scientific understanding of fugitive dust emissions and their estimates in emissions inventories.

1.2 Hypotheses for Discrepancies

The following hypotheses have been advanced as potential causes of discrepancies.

1. Lack of accounting for secondary aerosol contributions in estimating fraction of fugitive dust contributions to ambient PM_{10} and $PM_{2.5}$. Twenty percent of PM_{10}, and more than half of $PM_{2.5}$, may be composed of secondary sulfates or nitrates that are not part of the primary particle emissions inventories. These contributions need to be quantified and subtracted from ambient concentrations

prior to estimating the proportion of primary fugitive dust particles that are compared with emissions inventories.

2. Lack of accounting for global and regional crustal contributions to urban concentrations. Some of the concentrations of geological material measured in ambient air may result from long-range transport of sources outside of the United States or multi-day carry-over of very small dust particles. Lack of accounting for these emissions could result in higher dust concentrations measured in ambient air than would be estimated by modeling of a fugitive dust emissions inventory. If this hypothesis were true, the discrepancies between emissions inventories and ambient concentrations would be larger.

3. Incompatible temporal and spatial averaging of fugitive dust emissions relative to ambient PM measurements. Fugitive dust emission rates are often averaged over citywide or countywide areas and over a season or a year. Ambient urban dust contributions are often dominated by the presence of, or lack of, emissions from nearby sources.

4. Inaccurate formulation of emission factors. Emission factors for all fugitive dust emission source types are determined empirically from tests of a relatively small number of representative sources. The number and nature of these tests is small compared to the variability to be found in many different emission types and different geographical settings. Many of the current emission factors for PM_{10} and $PM_{2.5}$ were derived from measures of total suspended particulate (TSP), rather than from direct measurement of these size fractions. If the sources tested and the particle size modifiers applied in current emission factors do not represent the sources in an inventory, then estimates of particle emissions may be higher or lower than their actual values.

5. Insufficient and uncertain activity levels, specifically with respect to the reservoir of suspendable particles, particle size distributions in the reservoir, meteorological variables, and human intervention. Most emissions inventories assume replenishment of particle reservoirs between emission events and a minimal effectiveness of emission reduction strategies. These assumptions would result in estimated emissions being higher than actual values. Emissions inventories also apply land use and traffic data sets that may have changed for the year of record. These changes could result in positive or negative biases in emission estimates. Specific activity data that correspond to PM_{10} or $PM_{2.5}$ samples on a specific day are seldom available. Only long-term averages or statistical distributions of dust-generating activities are technically practical.

6. Insufficient accounting for injection heights, deposition losses, and horizontal impaction losses in dispersion models. Most fugitive dust particles are larger than 2 μm in aerodynamic diameter and deposit rapidly to the ground after suspension. If the majority of particles do not attain injection heights above a few meters above ground level they probably settle to the surface within a few minutes after release. Particles entrained in airflows that encounter vertical-standing trees,

shrubs, and buildings may impact on these surfaces, thereby removing them from the atmosphere. Under these conditions, suspended dust may cause neighborhood-scale "hot spots," but much of it will be removed before contributing to urban- or regional-scale PM_{10} or $PM_{2.5}$ concentrations. "Effective" emission rates from fugitive dust sources would be lower than current estimates if this hypothesis were true.

1.3 Structure of Report

Section 1 has stated the objectives of this study and defined the hypotheses to be tested. Section 2 summarizes national emission estimates and compares the fractions from fugitive dust with those determined from source apportionment studies of chemically characterized ambient samples. Geological contributions at background sites are identified and reasons are advanced for their sources. This information allows the first three hypotheses to be examined.

Section 3 summarizes the mechanisms that create emissions from unpaved roads, paved roads, soil disturbance and removal, and urban wind erosion. These are the major sources of fugitive dust in emissions inventories. Emission estimation methods applied to national inventories and to the state of California are contrasted and examined with respect to their representation of the physical processes. This information allows hypotheses three, four, and five to be examined.

Section 4 describes the emissions, chemical/transport, and receptor models that are applied, or that can be applied, to estimate fugitive dust contributions. This information allows the fifth and sixth hypotheses to be examined from the standpoint of dynamically modeling dust emissions for specific episodes.

Section 5 summarizes the conclusions from this review and recommends short-term and long-term research projects. This report is not a comprehensive review of fugitive dust emissions and ambient concentrations. It provides examples of current knowledge rather than an exhaustive or critical review of that knowledge. Such reviews are necessary and are defined as projects for future research. Citations are documented in Section 6. To facilitate this research, a comprehensive bibliography of reports and publications related to fugitive dust has been assembled in Section 7.

2. FUGITIVE DUST AND AIR POLLUTION

This section describes the nature of the discrepancies between emissions inventories and ambient particles. It examines the size distribution of particles that are usually found in the atmosphere and the portions of size distributions that are occupied by fugitive dust contributions. Annual emission estimates of PM_{10}, $PM_{2.5}$, and particle precursor gases are summarized from the U.S. National Trends Inventory. Chemical and physical characteristics of source emissions are examined to demonstrate how fugitive dust contributions can be distinguished from other sources by chemical analysis of ambient samples. Fugitive dust source contributions and fractions of primary PM_{10} from several receptor model source apportionment studies are compared with emissions inventory estimates. Background concentrations and fractions of PM_{10} and $PM_{2.5}$ are described and their potential U.S. and global sources are identified. This information is evaluated with respect to hypotheses about accounting for secondary aerosol in ambient samples, contributions from uninventoried background concentrations, and comparability of temporal and spatial scales.

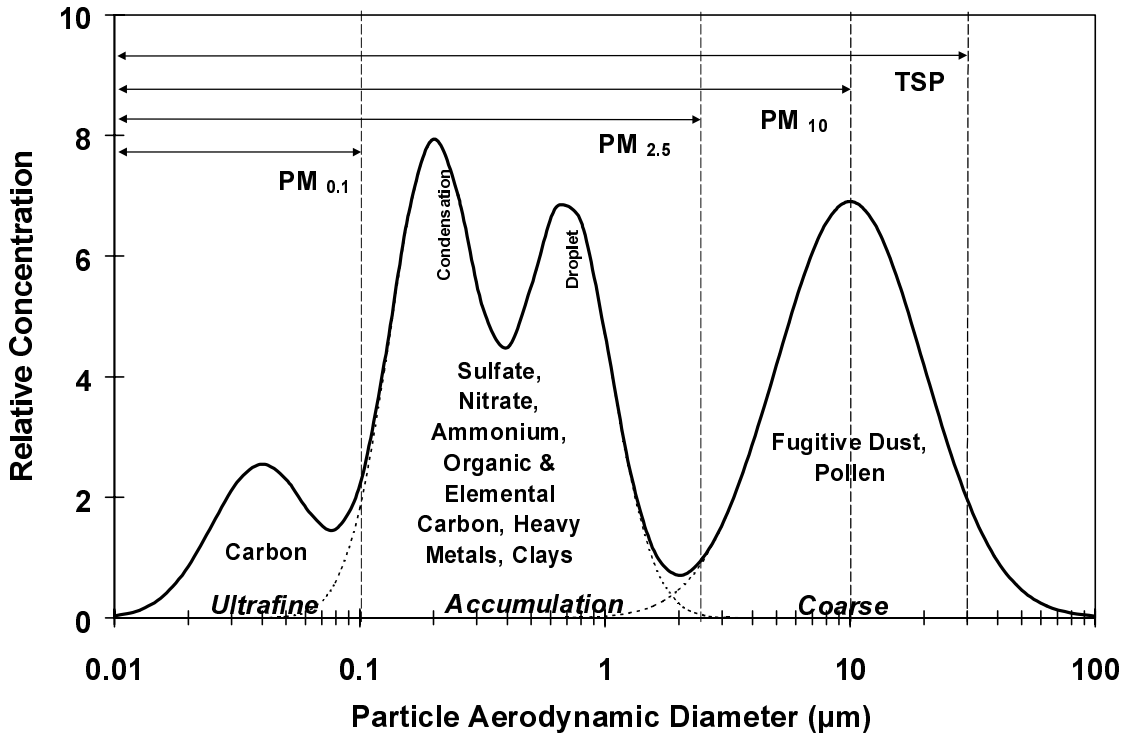
2.1 Particle Sizes

Whitby et al. (1972) described the major features of particle size mass distributions, illustrated in Figure 2-1; these features have been confirmed by many other investigators in many areas.

Most of the particles larger than ~2 or 3 μm are called "coarse particles" that result from grinding activities and are dominated by material of geological origin. Fugitive dust dominates this size range. Pollen and spores also inhabit the coarse particle size range, as do ground up trash, leaves, and tires. Figure 2-1 shows that this size range has a tail that overlaps with other size ranges, extending down to below 1 μm. Particles at the low end of the coarse size range also occur when cloud and fog droplets form in a polluted environment, then dry out after having scavenged other particles and gases (Jacob et al., 1986). Particles larger than 30 μm deposit to the surface within less than an hour after suspension unless they are injected to high altitudes. This deposition effectively limits atmospheric concentrations for very large particles. The peak of the coarse mode may shift between ~6 and 25 μm (Lundgren and Burton, 1995). This peak shifts toward larger particle sizes when fugitive dust emissions are close to the measurement location, and it shifts toward smaller particles at non-urban locations far from freshly generated dust emissions.

The "ultrafine particles" (Oberdörster et al., 1995; Kotzick et al., 1997) or "nucleation mode" in Figure 2-1, consists of particles with diameters less than ~0.08 μm that are emitted directly from combustion sources or that condense from cooled gases soon after emission. Ultrafine particle lifetimes are usually less than one hour because they rapidly coagulate with themselves or larger particles or serve as nuclei for cloud or fog droplets. The nucleation range is detected only when fresh emissions are close to a measurement site or when new particles have been recently formed in the atmosphere (Lundgren and Burton, 1995).

Figure 2-1. Typical distribution of particle sizes found in the atmosphere. Fugitive dust dominates the coarse mode of this distribution.



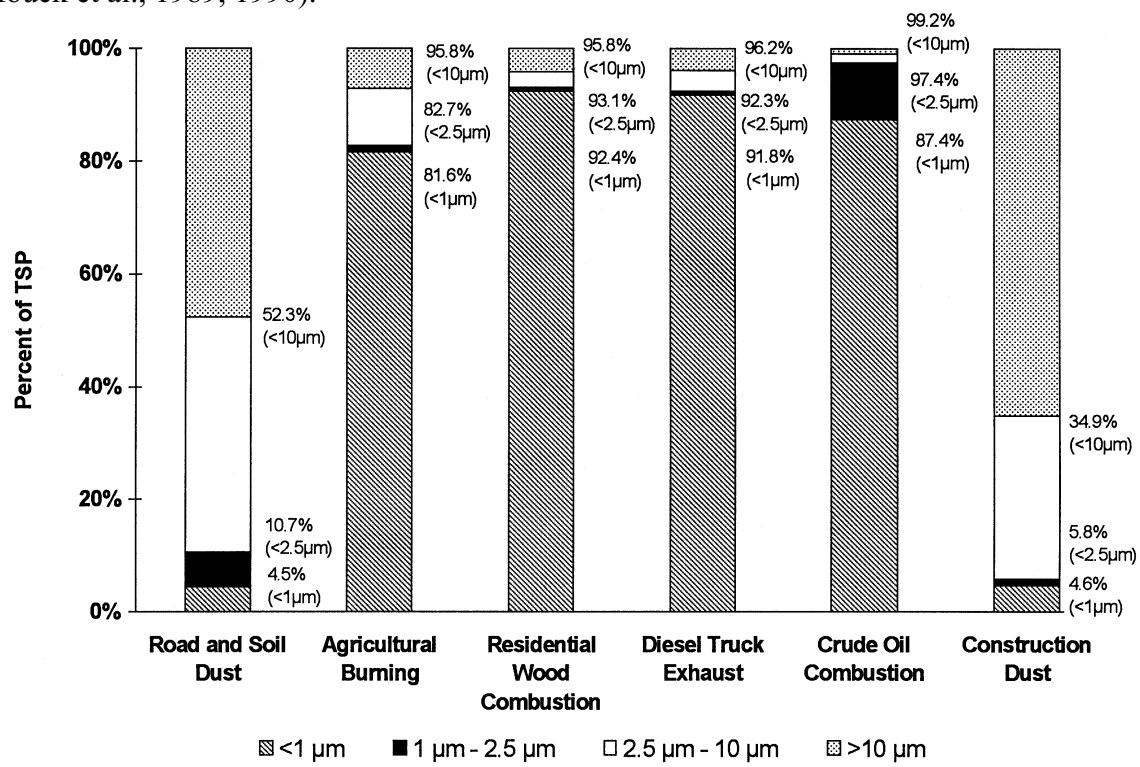
The "accumulation" range consists of particles with diameters between 0.08 and ~2 µm. These particles result from the coagulation of smaller particles emitted from combustion sources, from gas-to-particle conversion, from condensation of volatile species, and from finely ground dust particles. Chemical-specific size distributions show that these sub-modes exist in several different environments (Hering and Friedlander, 1982; Hoppel et al., 1990; Sloane et al., 1991). John et al. (1990) interpreted the peak centered at ~0.2 µm as a "condensation" mode containing gas-phase reaction products, and the ~0.7 µm peak as a "droplet" mode resulting from growth by nucleation of particles in the smaller size ranges and by reactions that take place in water droplets. The liquid water content of ammonium nitrate, ammonium sulfate, sodium chloride, and other soluble species increases with relative humidity, and this is especially important when relative humidity exceeds 70%. When these modes contain soluble particles, their peaks shift toward larger diameters as humidity increases (Tang, 1976, 1980, 1993; Tang et al., 1977; McMurry et al., 1987). The ultrafine and accumulation ranges constitute the "fine" particle size fraction; the majority of sulfuric acid, ammonium bisulfate, ammonium sulfate, ammonium nitrate, organic carbon, and elemental carbon is found in this size range.

The $PM_{2.5}$, PM_{10}, and TSP size fractions commonly measured by air quality monitors are identified in Figure 2-1 by the portion of the size spectrum that they occupy. The mass

collected is proportional to the area under the distribution within each size range. The TSP size fraction ranges from 0 to ~30 μm, the PM_{10} fraction ranges from 0 to 10 μm, and the $PM_{2.5}$ size fraction ranges from 0 to 2.5 μm in aerodynamic diameter. No sampling device operates as a step function, passing 100% of all particles below a certain size and excluding 100% of the particles larger than that size. When sampled, each of these size ranges contains a certain abundance of particles above the upper size designation of each range (Watson et al., 1983; Wedding and Carney, 1983). A small shift in the 50% cut-point of a PM_{10} sampler has a large influence on the mass collected because the coarse mode usually peaks near 10 μm. On the other hand, a similar shift in cut-point near 2.5 μm results in a small effect on the mass collected owing to the low quantities of particles in the 1 to 3 μm size range (Chow, 1995; Watson et al., 1995).

Figure 2-2 shows the size distribution of suspended particles measured from common emission sources. Construction dusts, road dusts, and soil dusts formed from pulverization of larger soil particles are predominantly in the coarse particle size range, with minor to moderate quantities in the $PM_{2.5}$ fraction. Combustion particles, on the other hand, dominate the $PM_{2.5}$ size fraction. Chemical components that distinguish between geological dusts, combustion products, and secondary aerosols can be used to effectively classify TSP or PM_{10} mass concentrations into accumulation or coarse fractions of the particle size distribution.

Figure 2-2. Size distributions of several particulate source emissions (Ahuja et al. 1989; Houck et al., 1989, 1990).



Based on average daily emissions in California's South Coast Air Basin (SoCAB), Countess (1999) reported $PM_{2.5}/PM_{10}$ emission ratios greater than 0.90 for most stationary sources and unplanned fires (with the exception of industrial process sources where $PM_{2.5}/PM_{10}$ = 0.56). $PM_{2.5}/PM_{10}$ emission ratios exceeded 0.98 for mobile sources and commercial charbroiling. $PM_{2.5}/PM_{10}$ ratios were in the range of 0.15 to 0.25 for fugitive dust sources (Cowherd and Kuykendal, 1997). Fugitive dust accounted for 80% of primary PM_{10} emissions, and for 48% of primary $PM_{2.5}$ emissions in the SoCAB. Estimated $PM_{2.5}$ contributions from secondary aerosols were twice those of primary particles in the SoCAB.

2.2 Emission Rate Estimates

Table 2-1 summarizes emission rate estimates for the United States during 1997. The PM_{10} and $PM_{2.5}$ fugitive dust portions of this national inventory consists of "Natural" wind erosion, "Miscellaneous" sources in "Agriculture and Forestry" and a general category called "Fugitive Dust," and the "Other Industrial Processes" of "Mineral Products". Also listed in Table 2-1 are emission rates for sulfur dioxide (SO_2), oxides of nitrogen (NO_x), and ammonia (NH_3) that are precursors to secondary sulfates ($SO_4^=$) and nitrates (NO_3^-).

The PM_{10} and $PM_{2.5}$ columns represent "primary" particles that are directly emitted from the identified sources. Of these primary emissions, fugitive dust emissions constitute 89% of the 33,574 thousand short tons per year (tpy) of PM_{10} and 66% of the 8,288 thousand tpy of $PM_{2.5}$. In contrast, on-road and off-road transportation account for 2.4%, open and residential vegetative burning accounts for 1.8%, and the other emitters (mostly industrial and residential coal, oil, and natural gas combustion) account for ~7% of total PM_{10} emissions. For $PM_{2.5}$ emissions, 7.5% derive from on-road and off-road vehicle exhaust, 6.9% come from open and residential vegetative burning, with the rest (~19%) deriving from the other emitters. By itself, Table 2-1 gives the impression that fugitive dust should be the major focus of control strategies to achieve PM_{10} and $PM_{2.5}$ air quality standards.

Sulfur dioxide (SO_2), oxides of nitrogen (NO_x), and ammonia (NH_3) are emitted as gases, but they transform into "secondary" particles in the atmosphere that add to the primary particle portions of PM_{10} and $PM_{2.5}$. The resulting particulate sulfates and nitrates are almost exclusively in the accumulation size range illustrated in Figure 2-1; their concentrations in $PM_{2.5}$ are only slightly less than their PM_{10} concentrations. Table 2-1 shows that national SO_2, and NO_x emissions are approximately three times primary $PM_{2.5}$ emissions. Even though only a fraction of these emissions (5% to 50%, depending on atmospheric conditions) changes to particles in the atmosphere, the equivalent particle emissions are equivalent to or higher than primary emissions from many of the listed sources.

Volatile Organic Compounds (VOC) also participate in the formation of secondary sulfates, nitrates, and organic particles. U.S. EPA (1998a) reports nearly 20,000 thousand tpy of VOCs emitted in 1997, many of them originating from the same sources listed in Table 2-1. Light hydrocarbons (containing fewer than 8 carbon atoms) constitute the majority of VOC emissions. These do not convert into particles, but they do participate in the photochemical reactions that oxidize SO_2 and NO_x to sulfates and nitrates. A portion of

Table 2-1. U.S. national emissions for 1997 (U.S. EPA, 1998a).

Category	Type	Thousands of tons per year (tpy)				
		PM_{10}	$PM_{2.5}$	NO_x	SO_2	NH_4
NATURAL SOURCES	Biogenic					18.4
	Geogenic - wind erosion	5315.8	797.4			
NATURAL SOURCES Total		5315.8	797.4			18.4
MISCELLANEOUS	Agriculture & Forestry	4707.2	927.4			2498.0
	Cooling Towers	1.4	1.4	0.0		
	Fugitive Dust	19429.5	3460.5	1.4	0.3	0.0
	Health Services			0.1		
	Other Combustion	1015.4	884.9	344.3	12.9	
MISCELLANEOUS Total		25153.5	5274.2	345.7	13.2	2498.0
FUEL COMB. ELEC. UTIL.	Coal	264.5	133.5	5598.6	12531.1	0.3
	Gas	0.5	0.5	288.5	4.3	3.8
	Internal Combustion	19.4	19.4	159.2	61.0	0.1
	Oil	5.7	5.0	132.2	485.6	1.9
FUEL COMB. ELEC. UTIL. Total		290.1	158.3	6178.4	13081.9	6.1
FUEL COMB. INDUSTRIAL	Coal	72.0	25.4	613.8	1768.6	0.0
	Gas	47.2	45.9	1384.5	571.7	13.7
	Internal Combustion	68.2	50.9	901.9	24.0	0.0
	Oil	48.2	28.3	240.1	847.2	4.0
	Other	78.0	62.2	129.9	153.4	0.0
FUEL COMB. INDUSTRIAL Total		313.5	212.6	3270.2	3364.8	17.7
FUEL COMB. OTHER	Commercial/Inst Coal	15.6	6.1	39.8	205.9	0.0
	Commercial/Inst Gas	6.7	6.3	241.2	8.1	0.8
	Commercial/Inst Oil	13.3	5.6	106.7	414.1	2.2
	Misc. Fuel Comb. (Except Residential)	74.0	73.5	29.8	5.6	
	Residential Other	19.5	16.2	824.5	174.0	5.2
	Residential Wood	368.3	368.3	33.7	4.8	
FUEL COMB. OTHER Total		497.4	476.0	1275.7	812.5	8.2
ON-ROAD VEHICLES	Diesels	156.4	143.9	1932.4	83.9	4.4
	Heavy-Duty Gas Vehicles	8.6	5.7	326.2	11.3	2.6
	Light-Duty Gas Trucks	40.3	25.1	1901.3	96.0	72.6
	Light-Duty Gas Vehicles & Motorcycles	55.6	32.3	2875.2	128.6	160.7
ON-ROAD VEHICLES Total		261.0	207.0	7035.2	319.7	240.3
NON-ROAD ENGINES AND VEHICLES	Aircraft	40.9	28.9	177.5	11.9	
	Marine Vessels	30.9	21.9	234.7	244.6	1.2

Table 2-1. (continued)

Category	Type	Thousands of tons per year (tpy)				
		PM_{10}	$PM_{2.5}$	NO_x	SO_2	NH_4
	Non-Road Diesel	315.7	290.4	2986.7	682.2	
	Non-Road Gasoline	51.0	43.0	211.4	7.1	
	Railroads	27.3	25.1	949.5	114.1	1.7
NON-ROAD ENGINES AND VEHICLES Total		465.8	409.3	4559.8	1059.9	2.9
OTHER INDUSTRIAL PROCESSES	Agriculture, Food, & Kindred Products	83.4	42.5	6.2	3.1	2.1
	Construction	0.0	0.0			
	Electronic Equipment	0.0	0.0	0.1	0.0	
	Machinery Products	7.3	3.0	6.9	0.6	
	Mineral Products	326.0	138.0	302.9	297.3	0.0
	Miscellaneous Industrial Processes	23.7		10.3	4.1	40.9
	Rubber & Miscellaneous Plastic Products	3.3		0.2	0.1	
	Textiles, Leather, & Apparel Products	0.3	0.2	0.1	0.1	
	Transportation Equipment	0.4	0.2	0.2		
	Wood, Pulp & Paper, & Publishing Products	85.5	66.7	93.9	121.7	
OTHER INDUSTRIAL PROCESSES Total		529.9	250.7	420.9	426.8	43.1
METALS PROCESSING	Ferrous Metals Processing	154.8	96.0	85.7	156.1	6.1
	Metals Processing NEC	22.7	16.8	3.8	18.6	0.0
	Nonferrous Metals Processing	42.3	26.4	12.6	377.6	0.0
METALS PROCESSING Total		219.9	139.2	102.1	552.2	6.1
PETROLEUM & RELATED INDUSTRIES	Asphalt Manufacturing	18.5		5.1	9.3	
	Oil & Gas Production	1.8	1.5	59.9	92.7	0.0
	Petroleum Refineries & Related Industries	21.0	13.7	49.7	282.6	44.7
PETROLEUM & RELATED INDUSTRIES Total		41.4	15.2	114.6	384.6	44.7
SOLVENT UTILIZATION	Degreasing	0.1	0.1	0.3	0.0	
	Dry Cleaning	0.0	0.0	0.0	0.0	
	Graphic Arts	0.4	0.3	0.7	0.1	
	Nonindustrial			0.0		
	Other Industrial	1.1	0.9	0.2	0.1	
	Solvent Utilization NEC			0.0		
	Surface Coating	4.8	4.2	1.8	0.5	

Table 2-1. (continued)

Category	Type	Thousands of tons per year (tpy)				
		PM_{10}	$PM_{2.5}$	NO_x	SO_2	NH_4
SOLVENT UTILIZATION Total		6.3	5.6	3.0	0.7	
STORAGE & TRANSPORT	Bulk Materials Storage	111.5	42.8	0.9	1.0	0.0
	Bulk Materials Transport	0.5	0.0			
	Bulk Terminals & Plants	0.0	0.0	0.6	0.2	
	Inorganic Chemical Storage	0.6	0.4	0.3	0.3	
	Organic Chemical Storage	0.6	0.5	3.8	0.1	
	Petroleum & Petroleum Product Storage	0.3	0.3	0.4	0.5	
	Petroleum & Petroleum Product Transport	0.0	0.0	0.3	0.1	
	Service Stations: Stage II	0.0	0.0	0.0	0.0	
STORAGE & TRANSPORT Total		113.6	44.1	6.4	2.1	0.0
CHEMICAL & ALLIED PRODUCT MFG	Agricultural Chemical Mfg	10.6	8.0	78.4	4.7	192.8
	Inorganic Chemical Mfg	4.8	3.6	7.3	208.0	
	Organic Chemical Mfg	30.6	11.9	20.8	8.9	
	Other Chemical Mfg	19.2	16.7	56.2	78.1	
	Paint, Varnish, Lacquer, Enamel Mfg	1.0	0.5	0.0	0.0	
	Pharmaceutical Mfg	0.2	0.1	0.0	0.3	
	Polymer & Resin Mfg	4.0	3.3	4.0	0.5	
CHEMICAL & ALLIED PRODUCT MFG Total		70.2	44.1	166.7	300.5	192.8
WASTE DISPOSAL & RECYCLING	Incineration	74.4	53.1	56.4	37.0	
	Industrial Waste Water	0.0	0.0	0.0	0.5	
	Landfills	0.5	0.5	0.7	0.1	
	Open Burning	220.3	200.3	45.5	11.4	
	Other	0.5	0.5	0.8	0.5	
	POTW	0.0	0.0	0.1	0.1	99.7
WASTE DISPOSAL & RECYCLING Total		295.7	254.4	103.5	49.7	99.7
Grand Total		33574.1	8288.2	23582.2	20368.7	3178.0

the heavy hydrocarbons (containing >8 carbon atoms) can change to secondary organic carbon particles under some conditions, but this is believed to constitute a small fraction of annual average organic carbon concentrations in most urban settings (Pandis et al., 1992, 1993).

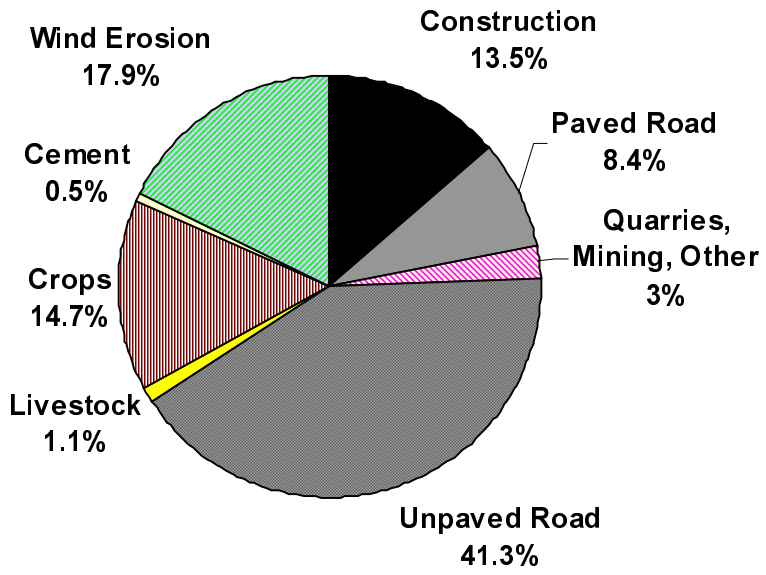
Although fugitive dust is predominantly a primary pollutant, it can play a minor role in secondary particle formation. Some components of dust, such as ammonium nitrate fertilizer, may volatilize into ammonia and nitric acid gases, thereby contributing to secondary ammonium nitrate aerosol (Mozurkewich, 1993; Watson et al., 1994a). Alkaline particles, such as calcium carbonate, may react with nitric and hydrochloric acid gases while on the ground, in the atmosphere, or on filter samples to form coarse particle nitrates and chlorides (Zhuang et al., 1999; Song and Carmichael, 1999). Ammonium sulfate and ammonium nitrate fertilizers and minerals such as gypsum (calcium sulfate) may be mistaken for secondary sulfates and nitrates.

Figure 2-3 shows the emissions from different fugitive dust sub-types (U.S. EPA, 1998a). Industrial emissions from cement production, quarrying, mining, and other emitters are a small fraction of the total. Unpaved roads, paved roads, construction, and wind erosion together constitute more than 80% of PM_{10} dust emissions and 75% of $PM_{2.5}$ emissions. These categories are from both urban and non-urban areas. Nearly all of the crop and livestock emissions are from non-urban areas. Most of the paved road dust and construction emissions are from urban areas. Unpaved road dust emissions are from both urban and non-urban sources, with a higher proportion from non-urban surroundings. Wind erosion is primarily non-urban. These breakdowns give the further impression that unpaved road dust is the most important urban fugitive dust contributor on a nationwide basis.

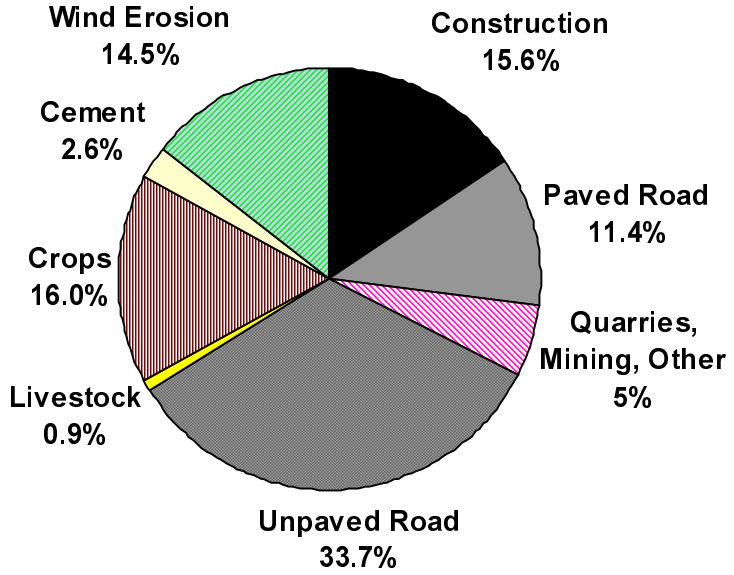
Emission rates in Table 2-1 are calculated by multiplying an emission factor for each source type by an activity level. Emission factors, described in Section 3, quantify the weight of emissions within a size range per unit of activity (U.S. EPA, 1998b). These factors sometimes include modifiers for the appropriate particle size fraction (TSP, PM_{10}, or $PM_{2.5}$), the size of the dust reservoir (i.e., surface loadings), and effects of pollution controls (e.g., dust suppressants or dust removal). Activity databases include statewide estimates of paved and unpaved roadway miles, vehicle miles traveled, dollars spent on construction, acres under cultivation, vegetation coverage, annual precipitation, and other meteorological conditions. Emissions are further allocated to counties based on population, land area, or other available indicators.

Each of the components necessary to compile an inventory contains random and systematic uncertainties. Random errors are associated with the inherent variability in the process or processes that cause emissions. These errors are typically referred to as variability and are normally described by ranges or confidence intervals. Even if all other sources of uncertainty are removed, the variability remains. Some processes are inherently more variable than others, resulting in larger uncertainties. Systematic errors cause a positive or negative bias that exceeds the random variability. These biases were described in the Section 1 hyptotheses.

Figure 2-3. Fraction of PM_{10} and $PM_{2.5}$ fugitive emissions for unpaved roads, paved roads, construction, wind erosion, crops, livestock, and industrial sources. Fugitive dust constitutes 88% of total PM_{10} and 66% of $PM_{2.5}$ emissions.



PM_{10} Fugitive Dust=29,778 of 33,574 thousand tpy



$PM_{2.5}$ Fugitive Dust=5,511 of 8,288 thousand tpy

Fugitive dust emission estimates contain a high amount of variability owing to the meteorological, physical, and chemical factors on which emissions are based. These factors can vary widely on a national or smaller regional basis. While researchers rarely consider activity levels to be one to two orders of magnitude too high or low, they can be biased in either direction by factors of two to five. Averaging over an entire year and all of the United States probably eliminates much of the random error. Systematic errors probably dominate over random errors for spatially and temporally aggregated emission rates. Random error is much more important for smaller areas and shorter time intervals, such as those used for modeling pollution episodes.

State and local inventories often have access to more precise activity levels, and they may have developed area specific emission factors. For this reason there may be discrepancies between the national U.S. inventory and state or local emission estimates. The basic features are the same, however. Fugitive dust sources dominate the primary PM_{10} and $PM_{2.5}$ emission rates for all types of primary particle emissions inventories.

2.3 Chemical Composition of Fugitive Dust and Other Sources

Geological contributions to PM_{10} and $PM_{2.5}$ concentrations in ambient air are easily distinguished from other source contributions by their chemical profiles. Chemical source profiles are the fractional mass abundances of measured chemical species relative to primary particulate matter (PM) mass in source emissions. Hundreds of fugitive dust source profiles from many different areas have been measured (Chow and Watson, 1994a).

Source profile compilations (e.g., Watson, 1979; Sheffield and Gordon, 1986; Core and Houck, 1987; Cooper et al., 1987; Chow et al., 1989; Houck et al., 1989a, 1989b, 1989c, 1989d; Shareef et al., 1989; Watson et al., 1990a; Chow and Watson, 1994a, Watson et al., 1994c, 1996a, Chow and Watson, 1997a, 1997b) include chemical abundances of elements, ions, and carbon for geological material (e.g., paved and unpaved road dust, soil dust, storage pile), motor vehicle exhaust (e.g., diesel-, leaded-gasoline-, and unleaded-gasoline-fueled vehicles), vegetative burning (e.g., wood stoves, fireplaces, forest fires, and prescribed burning), industrial boiler emissions, and other aerosol sources. More modern, research-oriented profiles include specific organic compounds or functional groups, elemental isotopes, and microscopic characteristics of single particles (Schauer et al., 1996; Zielinska et al., 1998). Table 2-2 summarizes the chemical components that are found in most source emissions.

Geological profiles typically show large abundances of aluminum (Al), silicon (Si), potassium (K), calcium (Ca), and iron (Fe). The abundance of total potassium (K) is six to ten times the abundance of soluble potassium (K^+). Al, Si, K, and Fe abundances are similar among geolotical profiles. Lead (Pb) is often enriched in paved road dust, even though leaded fuels are no longer used in the United States. Carbon constitutes 5% to 15% of road dust and some agricultural soils. Soluble ions such as sulfate, nitrate, and ammonium are generally low, in the range of 0.1% to 0.2%. Sodium (Na^+) and chloride (Cl^-) are also low, except for situations when salt is used as a de-icing agent.

Table 2-2. Chemical abundances in different source types.

Source Type	Dominant Particle Size[a]	Chemical Abundances in Percent Mass			
		< 0.1%	**0.1 to 1%**	**1 to 10%**	**> 10%**
Paved Road Dust	Coarse	Cr, Sr, Pb, Zr	$SO_4^{=}$, Na^+, K^+, P, S, Cl, Mn, Zn, Ba, Ti	Elemental Carbon (EC), Al, K, Ca, Fe	Organic Carbon (OC), Si
Unpaved Road Dust	Coarse	NO_3^-, NH_4^+, P, Zn, Sr, Ba	$SO_4^{=}$, Na^+, K^+, P, S, Cl, Mn, Ba, Ti	OC, Al, K, Ca, Fe	Si
Construction	Coarse	Cr, Mn, Zn, Sr, Ba	$SO_4^{=}$, K^+, S, Ti,	OC, Al, K, Ca, Fe	Si
Agricultural Soil	Coarse	NO_3^-, NH_4^+, Cr, Zn, Sr	$SO_4^{=}$, Na^+, K^+, S, Cl, Mn, Ba, Ti	OC, Al, K, Ca, Fe	Si
Natural Soil	Coarse	Cr, Mn, Sr, Zn, Ba	Cl^-, Na^+, EC, P, S, Cl, Ti	OC, Al, Mg, K, Ca, Fe	Si
Lake Bed	Coarse	Mn, Sr, Ba	K^+, Ti	$SO_4^{=}$, Na^+, OC, Al, S, Cl, K, Ca, Fe	Si
Motor Vehicle	Fine	Cr, Ni, Y	NH_4^+, Si, Cl, Al, Si, P, Ca, Mn, Fe, Zn, Br, Pb	Cl^-, NO_3^-, $SO_4^{=}$, NH_4^+, S	OC, EC
Vegetative Burning	Fine	Ca, Mn, Fe, Zn, Br, Rb, Pb	NO_3^-, $SO_4^{=}$, NH_4^+, Na^+, S	Cl^-, K^+, Cl, K	OC, EC
Residual Oil Combustion	Fine	K^+, OC, Cl, Ti, Cr, Co, Ga, Se	NH_4^+, Na^+, Zn, Fe, Si	V, OC, EC, Ni	S, $SO_4^{=}$
Incinerator	Fine	V, Mn, Cu, Ag, Sn	K^+, Al, Ti, Zn, Hg	NO_3^-, Na^+, EC, Si, S, Ca, Fe, Br, La, Pb	$SO_4^{=}$, NH_4^+, OC, Cl
Coal-Fired Boiler	Fine	Cl, Cr, Mn, Ga, As, Se, Br, Rb, Zr	NH_4^+, P, K, Ti, V, Ni, Zn, Sr, Ba, Pb	$SO_4^{=}$, OC, EC, Al, S, Ca, Fe	Si
Oil-Fired Power Plant	Fine	V, Ni, Se, As, Br, Ba	Al, Si, P, K, Zn	NH_4^+, OC, EC, Na, Ca, Pb	S, $SO_4^{=}$
Steel Blast Furnace	Fine	V, Ni, Se,	Al, Si, P, K, Zn	Mn, OC, EC	Fe
Smelter Fire	Fine	V, Mn, Sb, Cr, Ti	Cd, Zn, Mg, Na, Ca, K, Se	Fe, Cu, As, Pb	S
Antimony Roaster	Fine	V, Cl, Ni, Mn	$SO_4^{=}$, Sb, Pb	S	None reported
Marine	Fine and Coarse	Ti, V, Ni, Sr, Zr, Pd, Ag, Sn, Sb, Pb	Al, Si, K, Ca, Fe, Cu, Zn, Ba, La	NO_3^-, $SO_4^{=}$, OC, EC	Cl^-, Na^+, Na, Cl

[a] Coarse particles are between 2.5 and 10 µm in diameter; fine particles are less than 2.5 µm in diameter.

Vehicle exhaust and burning, other major emitters in Table 2-1, are constituted mostly by carbonaceous material with elemental levels much less than 1% of mass emissions. Soluble potassium to total potassium (K^+/K) ratios (Calloway et al., 1989) are typically 0.80 to 0.90 in vegetative burning profiles, in contrast to the low ratios found in geological material.

Ducted industrial source emissions often contain many different elements, as shown in Table 2-2. Primary particles emitted by coal-fired power generators have the largest overlap in composition with fugitive dust sources. Crustal elements such as Si, Ca, and Fe in coal-fired boiler profiles are present at 30% to 50% of the corresponding levels in geological material; Al abundances are equal to or higher than those found in geological material. Other elements such as phosphorus (P), K, titanium (Ti), chromium (Cr), manganese (Mn), strontium (Sr), zirconium (Zr), and barium (Ba) are present at less than 1% levels. Selenium (Se) is usually found in coal-fired boiler emissions that do not pass through dry sulfur dioxide scrubbers. Watson et al. (2000) found that limestone scrubbers processing hot exhaust gases removed selenium vapor before it could condense on fly ash particles.

As a group, fugitive sources are distinguishable from other types of sources on the basis of their crustal element abundances. Urban fugitive dust source profiles are often too similar to permit source resolution based on elemental abundances, but there are some distinct global and regional profiles. Gatz and Prospero (1998) applied Si/Al, Ca/Al ratios to separate transported North African dust from other sources. Gatz (1984) and Gatz et al. (1985) separated agricultural soil and unpaved roads based on their Ca/K ratios. Bruns et al. (1998) identify different microbial species in different types of agricultural soils. Houck et al. (1989c) showed clear distinctions in Na, Cl, $SO_4^=$, and carbonate ($CO_3^=$) abundances for alkaline lake beds relative to other sources. Davis and Chen (1993) identified clay and other specific minerals in a variety of soils. Freeman et al. (1991) used the gold (Au) abundance to separate crushed mining ore contributions from those of the overburden. Finding additional markers that further distinguish among fugitive dust emitters would improve emission estimation methods substantially.

2.4 Ambient PM_{10} and $PM_{2.5}$ Chemical Composition and Source Contributions

The relative abundances of chemical components in the atmosphere reflect the characteristics of emission sources. Major chemical components of $PM_{2.5}$ or PM_{10} mass in urban and non-urban areas consist of geological material, carbon, nitrate, sulfate, ammonium, carbon, sodium chloride, and liquid water:

- **Geological Material:** Suspended dust consists mainly of oxides of Al, Si, Ca, Ti, Fe, and other metals oxides (Taylor, 1964; Mason, 1966). The precise combination of these minerals depends on the geology of the area and industrial processes. Geological material typically constitutes ~50% of PM_{10} while only contributing 5 to 15% of $PM_{2.5}$.

- **Organic Carbon:** Particulate organic carbon consists of hundreds, possibly thousands, of separate compounds. The mass concentration of organic carbon can

be measured, as can carbonate carbon, but only about 10% of specific organic compounds that it contains have been measured. Vehicle exhaust (Rogge et al., 1993a; Zielinska et al., 1998), residential and agricultural burning (Rogge et al., 1998; Zielinska et al., 1998), meat cooking (Rogge et al., 1991; Zielinska et al., 1998), fuel combustion (Rogge et al., 1993b, 1997), road dust (Rogge et al., 1993c), and particle formation from heavy hydrocarbon (C_8 to C_{20}) gases (Pandis et al., 1992) are the major sources of organic carbon in $PM_{2.5}$. Because of this lack of molecular specificity, and owing to the semi-volatile nature of many carbon compounds, particulate "organic carbon" is operationally defined by the sampling and analysis method (Hering et al., 1984; Chow et al., 1993a, 2000).

- **Elemental Carbon:** Elemental carbon is black, often called "soot." Elemental carbon contains pure, graphitic carbon, but it also contains high molecular weight, dark-colored, non-volatile organic materials such as tar, biogenics, and coke. Elemental carbon usually accompanies organic carbon in combustion emissions with diesel exhaust (Watson et al., 1994c) being the largest contributor.

- **Nitrate:** Ammonium nitrate (NH_4NO_3) is the most abundant nitrate compound, resulting from a reversible gas/particle equilibrium between ammonia gas (NH_3), nitric acid gas (HNO_3), and particulate ammonium nitrate. Because this equilibrium is reversible, ammonium nitrate particles can easily evaporate in the atmosphere, or after they have been collected on a filter, owing to changes in temperature and relative humidity (Stelson and Seinfeld, 1982a, 1982b; Allen et al., 1989). Sodium nitrate ($NaNO_3$) is found in the $PM_{2.5}$ and coarse fractions near sea coasts and salt playas (Pilinus et al. 1987; Watson et al., 1994a) where nitric acid vapor irreversibly reacts with sea salt (NaCl).

- **Sulfate:** Ammonium sulfate ($(NH_4)_2SO_4$), ammonium bisulfate ((NH_4HSO_4), and sulfuric acid (H_2SO_4) are the most common forms of sulfate found in atmospheric particles, resulting from conversion of gases to particles. These compounds are water-soluble and reside almost exclusively in the $PM_{2.5}$ size fraction. Sodium sulfate (Na_2SO_4) may be found in coastal areas where sulfuric acid has been neutralized by sodium chloride (NaCl) in sea salt. Though gypsum ($CaSO_4$) and some other geological compounds contain sulfate, these are not easily dissolved in water for chemical analysis. They are more abundant in the coarse fraction than in $PM_{2.5}$ and are usually classified in the geological fraction.

- **Ammonium:** Ammonium sulfate, ammonium bisulfate, and ammonium nitrate (NH_4NO_3) are the most common compounds. The sulfate compounds result from irreversible reactions between sulfuric acid and ammonia gas, while the ammonium nitrate can migrate between gases and particle phases (Watson et al., 1994a). Ammonium ions may coexist with sulfate, nitrate, and hydrogen ions in small water droplets. While most of the sulfur dioxide and oxides of nitrogen precursors of these compounds originate from fuel combustion in stationary and mobile sources, most of the ammonia derives from living beings, especially animal husbandry practiced in dairies and feedlots.

- **Sodium Chloride:** Salt is found in suspended particles near sea coasts, open playas, and after de-icing materials are applied. Bulk sea water contains 57±7% chloride, 32±4% sodium, 8±1% sulfate, 1.1±0.1% soluble potassium, and 1.2±0.2% calcium (Pytkowicz and Kester, 1971). In its raw form (e.g., deicing sand), salt is usually in the coarse particle fraction and classified as a geological material (Chow et al., 1996a). After evaporating from a suspended water droplet (as in sea salt or when resuspended from melting snow), it is abundant in the $PM_{2.5}$ fraction. Sodium chloride is often neutralized by nitric or sulfuric acid in urban air where it is encountered as sodium nitrate or sodium sulfate (Pilinis et al., 1987).

- **Liquid Water:** Soluble nitrates, sulfates, ammonium, sodium, other inorganic ions, and some organic material absorb water vapor from the atmosphere, especially when relative humidity exceeds 70% (Tang and Munkelwitz, 1993; Saxena and Hildemann, 1997). Sulfuric acid absorbs some water at all humidities. Particles containing these compounds grow into the droplet mode as they take on liquid water. Some of this water is retained when particles are sampled and weighed for mass concentration. The precise amount of water quantified in a $PM_{2.5}$ depends on its ionic composition and the equilibration relative humidity applied prior to laboratory weighing.

Receptor models such as the Chemical Mass Balance (Watson et al., 1984, 1990b, 1990c, 1991, 1997a, 1998a) use the chemical compositions of particulate source emissions and of particles in ambient air to estimate the contributions of different sources to PM_{10} and $PM_{2.5}$ concentrations. Table 2-3 reanalyzes the summary of source contribution estimates in Watson et al. (1998a) by estimating the mass fraction of primary PM_{10} contributed by fugitive dust, motor vehicle exhaust, vegetative burning, and other source types. Source contributions from secondary ammonium sulfate and ammonium nitrate contributions have been subtracted from measured PM_{10} to estimate the concentration attributable only to primary emissions. Total and primary PM_{10} concentrations in the cited studies are tabulated in the third and fourth columns of Table 2-3.

Fugitive dust source contributions range from 12% to 99% of primary PM_{10} with most of the fractions being in the range of 40% to 60%. The tabulated studies are dominated by western sites where the PM_{10} standard was often exceeded, and fugitive dust was nearly always a major contributor. Geological contributions at the Ohio sites ranged from 20% to 45% of primary PM_{10}. Although source apportionment studies are not as extensive in the midwestern, southern, and eastern United States, it is expected that fugitive dust source contributions are lower owing to higher levels of vegetation on exposed surfaces.

Comparing total PM_{10} to primary PM_{10} in columns three and four of Table 2-3 shows that secondary sulfates and nitrates are often a large fraction of PM_{10} and an even larger component of $PM_{2.5}$. This is especially noticeable at the Rubidoux, Bakersfield, and Fresno, CA monitors. Subtracting these secondary components prior to estimating the fugitive dust fraction decreases, but does not eliminate, the discrepancy between ambient source

Table 2-3. Fractions of fugitive dust and other source contributions in primary PM_{10} from receptor model source apportionment studies.

		µg/m³		Percent of Primary PM_{10}			
Location	**Time Period**	**PM_{10}**	**Primary PM_{10}**	**Fugitive Dust**	**Exhaust**	**Burning**	**Other**
Central Phoenix, AZ (Chow et al., 1991)	Winter 1989-90	64.0	61.0	54.1	41.0	3.8	1.1
Corona de Tucson, AZ (Chow et al., 1992b)	Winter 1989-90	19.1	17.2	98.8	9.3	0.0	0.0
Craycroft, AZ (Chow et al., 1992b)	Winter 1989-90	23.4	22.1	58.8	37.6	0.0	3.6
Downtown Tucson, AZ (Chow et al., 1992b)	Winter 1989-90	48.0	46.8	66.5	29.9	0.0	3.6
Orange Grove, AZ (Chow et al., 1992b)	Winter 1989-90	34.2	33.4	59.9	44.9	0.0	0.0
Phoenix, AZ (Estrella Park) (Chow et al., 1991)	Winter 1989-90	55.0	53.4	69.3	18.7	1.7	10.3
Phoenix, AZ (Gunnery Rg.) (Chow et al., 1991)	Winter 1989-90	27.0	26.0	76.9	21.2	0.0	1.9
Phoenix, AX (Pinnacle Pk.) (Chow et al., 1991a)	Winter 1989-90	12.0	11.1	63.1	26.1	9.0	1.8
Rillito, AZ (Thanukos et al., 1992)	1988	79.5	79.5	71.1	1.5	0.0	27.4
Scottsdale, AZ (Chow et al., 1991)	Winter 1989-90	55.0	50.8	49.2	37.4	14.6	0.0
West Phoenix, AZ (Chow et al., 1991)	Winter 1989-90	69.0	65.5	45.8	38.2	15.3	0.8
Anacapa Island (Channel Islands) marine site (Chow et al., 1996b)	1989	26.0	21.0	12.9	7.1	0.0	58.9[a]
Anaheim, CA (Gray et al., 1988)	1986	52.1	35.3	60.1	11.6	0.0	28.3
Anaheim, CA (Summer) (Watson et al., 1994b)	Summer 1987	51.3	39.4	28.9	21.6	0.0	49.5
Anaheim, CA (Fall) (Watson et al., 1994b)	Fall 1987	104.0	61.8	21.4	60.2	0.0	18.4
Azusa, CA (Summer) (Watson et al., 1994b)	Summer 1987	92.1	74.6	46.8	21.3	0.0	31.9
Bakersfield, CA (Magliano, 1988)	1986	67.6	62.0	49.0	8.9	15.5	26.6
Bakerfield, CA (Chow et al., 1992c)	1988-89	79.6	61.4	72.5	12.5	10.6	4.4
Burbank, CA (Gray et al., 1988)	1986	56.6	39.2	54.3	15.6	0.0	30.1
Burbank, CA (Summer) (Watson et al., 1994b)	Summer 1987	72.3	53.4	26.2	31.8	0.0	41.9
Burbank, CA (Fall) (Watson et al., 1994b)	Fall 1987	94.8	66.6	16.5	58.7	0.0	24.8
Chula Vista 1, CA (Bayside) (Cooper et al., 1988)	1986	28.8	21.3	31.5	3.8	0.0	64.8
Chula Vista 2, CA (Del Ray) (Cooper et al., 1988)	1986	31.1	22.2	38.3	6.8	0.0	55.0
Chula Vista 3, CA (Cooper et al., 1988)	1986	29.6	21.4	46.7	6.5	0.0	46.7
Claremont, CA (Summer) (Watson et al., 1994b)	Summer 1987	70.0	54.2	35.8	26.6	0.0	37.6

Table 2-3. (continued)

		µg/m^3		Percent of Primary PM_{10}			
Location	**Time Period**	**PM_{10}**	**Primary PM_{10}**	**Fugitive Dust**	**Exhaust**	**Burning**	**Other**
Crows Landing, CA (Chow et al., 1992c)	1988-89	52.5	43.2	74.5	5.1	7.9	12.5
Downtown Los Angeles, CA (Gray et al., 1988)	1986	60.2	41.4	57.5	15.5	0.0	27.1
Downtown Los Angeles, CA (Summer) (Watson et al., 1994b)	Summer 1987	67.6	50.2	25.3	32.3	0.0	42.4
Downtown Los Angeles, CA (Fall) (Watson et al., 1994b)	Fall 1987	98.6	67.2	14.0	61.2	0.0	24.9
Fellows, CA (Chow et al., 1992c)	1988-89	54.6	42.0	72.4	5.0	8.1	14.5
Fresno, CA (Magliano, 1988)	1986	48.1	46.3	38.4	8.6	19.9	33.0
Fresno, CA (Chow et al., 1992c)	1988-89	71.5	57.5	55.3	11.8	8.9	24.0
Hawthorne, CA (Summer) (Watson et al., 1994b)	Summer 1987	45.9	30.3	24.8	18.5	0.0	56.8
Hawthorne, CA (Fall) (Watson et al., 1994b)	Fall 1987	85.1	59.6	14.9	58.9	0.0	26.2
Indio, CA (Kim et al., 1992)	1988-89	58.0	50.3	71.6	8.7	14.1	5.6
Kern Wildlife Refuge, CA (Chow et al., 1992c)	1988-89	47.8	43.0	39.8	5.1	9.3	45.8
Lennox, CA (Gray et al., 1988)	1986	46.9	31.4	51.3	14.3	0.0	34.4
Long Beach, CA (Gray et al., 1988)	1986	51.9	34.7	59.7	14.7	0.0	25.6
Long Beach, CA (Summer) (Watson et al., 1994b)	Summer 1987	46.1	34.4	32.3	18.3	0.0	49.4
Long Beach, CA (Fall) (Watson et al., 1994b)	Fall 1987	96.1	69.1	16.4	61.9	0.0	21.7
Magnolia, CA (Chow et al., 1992a)	1988	66.0	41.4	76.6	27.1	0.0	0.0
Palm Springs, CA (Kim et al., 1992)	1988-89	35.1	27.2	65.4	8.5	18.8	7.4
Riverside, CA (Chow et al., 1992a)	1988	64.0	37.8	86.2	18.5	0.0	0.0
Rubidoux, CA (Gray et al., 1988)	1986	87.4	59.7	78.9	9.4	0.0	11.7
Rubidoux, CA (Summer) (Watson et al., 1994b)	Summer 1987	114.8	77.9	50.6	22.2	0.0	27.2
Rubidoux, CA (Fall) (Watson et al., 1994b)	Fall 1987	112.0	78.3	45.1	38.7	0.0	16.2
Rubidoux, CA (Chow et al., 1992a)	1988	87.0	60.0	80.0	17.0	0.0	3.0
San Jose, CA (4th St.) (Chow et al., 1995)	Winter 1991/92	68.4	52.8	24.8	17.4	59.3	0.0
San Jose, CA (San Carlos St.) (Chow et al., 1995)	Winter 1991/92	64.9	50.0	23.6	17.8	62.6	0.0
San Nicolas Island, CA (Summer) (Watson et al., 1994b)	Summer 1987	17.4	13.2	12.1	6.8	0.0	81.1

Table 2-3. (continued)

		µg/m^3		Percent of Primary PM_{10}			
Location	**Time Period**	**PM_{10}**	**Primary PM_{10}**	**Fugitive Dust**	**Exhaust**	**Burning**	**Other**
Santa Barbara, CA (Chow et al., 1996b)	1989	34.0	29.8	31.9	49.3	0.0	18.8
Santa Barbara, CA (Gaviota Terminal) (Chow et al., 1996b)	1989	20.5	17.2	18.6	29.7	0.0	51.7
Santa Maria, CA (Chow et al., 1996b)	1989	27.0	22.5	32.9	33.8	0.0	33.3
Santa Ynez, CA (Chow et al., 1996b)	1989	19.0	16.2	28.4	42.0	0.0	29.6
Stockton, CA (Chow et al., 1992a)	1988-89	62.4	52.3	66.7	9.9	9.2	14.1
Upland, CA (Gray et al., 1988)	1986	58.0	37.1	69.5	11.1	0.0	19.4
Vandenberg AFB, CA (Watt Road) (Chow et al., 1996b)	1989	20.6	17.7	25.4	18.1	0.0	56.5
Telluride 1, CO (Central) (Dresser and Baird, 1988)	Winter 1986	208.0	208.0	15.4	0.0	47.5	37.2
Telluride 2, CO (Society Turn) (Dresser and Baird, 1988)	Winter 1986	27.0	27.0	44.8	0.0	27.0	28.1
Pocatello, ID (Houck et al., 1992)	1990	100.0	100.0	15.8	0.1	0.0	84.1
S. Chicago, IL (Hopke et al., 1988)	1986	80.1	64.7	45.7	4.3	0.0	49.9
S.E. Chicago, IL (Vermette et al., 1992)	1988	41.0	33.3	44.1	2.7	0.0	53.2
Reno, NV (Non-sweeping) (Chow et al., 1990)	Winter 1987	20.4	19.6	49.5	44.4	0.5	5.6
Reno, NV (Sweeping) (Chow et al., 1990)	Winter 1987	24.9	23.9	49.4	46.0	5.0	0.0
Reno, (Chow et al., 1988)	1986-87	30.0	28.1	53.0	35.6	6.8	4.6
Sparks, NV (Chow et al., 1988)	1986-87	41.0	37.4	40.4	31.0	35.8	0.0
Verdi, NV (Chow et al., 1988)	1986-87	15.0	14.0	55.7	28.6	7.9	7.9
Follansbee, OH (Skidmore and Chow, 1992)	1991	66.0	50.0	20.0	70.0	0.0	10.0
Mingo, OH (Skidmore and Chow, 1992)	1991	60.0	45.0	26.7	31.1	9.1	33.1
Sewage Plant, OH (Skidmore and Chow, 1992)	1991	62.0	49.0	44.9	24.5	0.0	30.6
Steubenville, OH (Skidmore and Chow, 1992)	1991	46.0	32.0	25.9	43.8	2.5	27.8
WTOV Tower, OH (Skidmore and Chow, 1992)	1991	49.0	34.0	21.8	47.1	0.6	30.6

[a] 55% was marine aerosol contributions.

contributions and emissions inventory estimates of source importance. Other emitters, especially vehicle exhaust, contribute proportionately more to measured PM_{10} than is indicated by inventories, even when these inventories are specific to the areas being studied.

Receptor source contributions and emission estimates are not entirely comparable. The source contributions in Table 2-3 are specific to the sampling sites listed and can only be extrapolated to the larger areas represented by emissions inventories when those sites are shown to represent these large areas. Chow et al. (1992a) estimated fugitive dust contributions at Rubidoux, CA, to be twice those at sites that were only 5 km from this site, indicating substantial nearby fugitive dust emissions. More systematic, long-term source apportionment studies are needed for comparison with local emissions inventories. More chemically speciated $PM_{2.5}$ measurements in urban areas are also needed. The planned EPA $PM_{2.5}$ national speciation network will provide some of these measurements needed to generalize the source contributions in Table 2-3 to longer measurement periods and a larger number of locations.

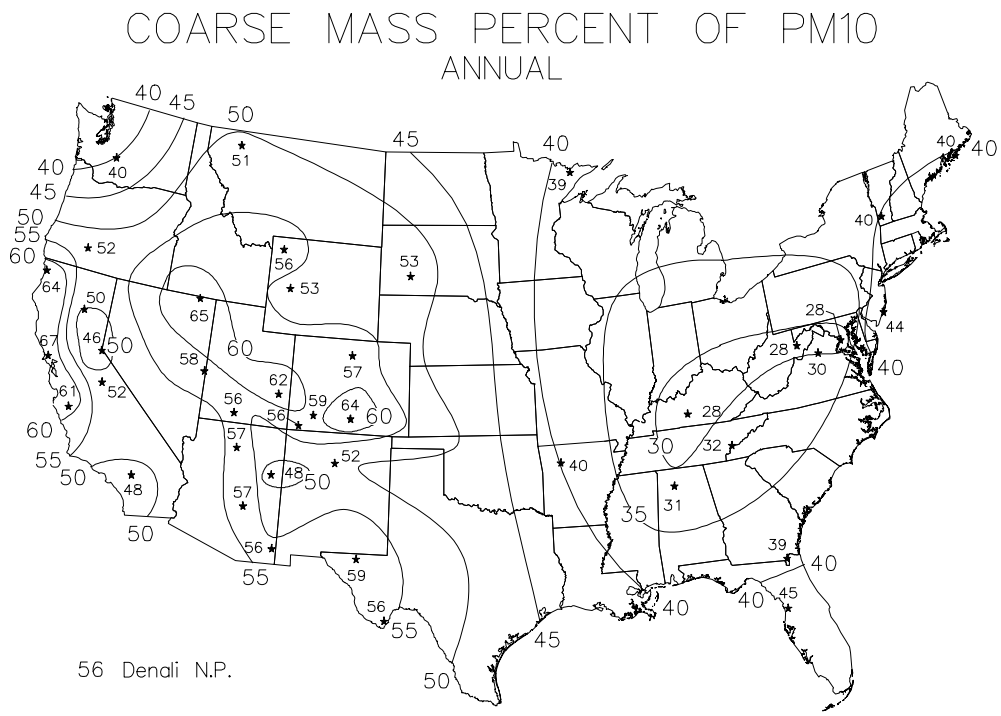
2.5 Background and Global Dust Contributions

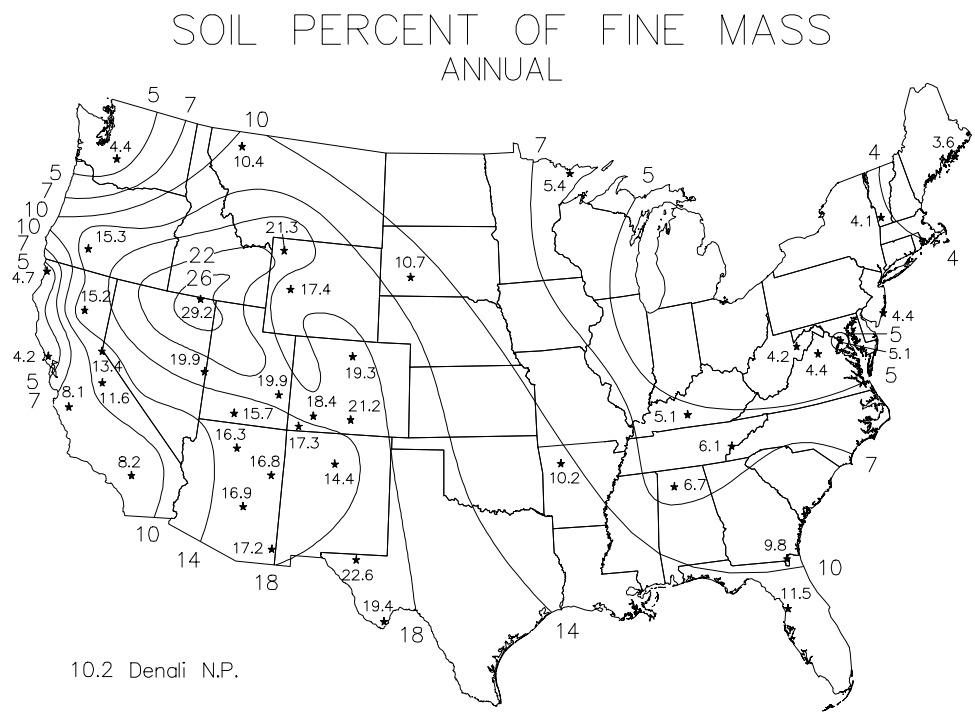
Geological material in the atmosphere is ubiquitous throughout the world, and a portion of the source contributions in Table 2-3 should be attributed to this background. Figure 2-4 from the Interagency Monitoring of PRotected Visual Environments (IMPROVE) visibility network provides an estimate of the fraction of PM_{10} and $PM_{2.5}$ constituted by dust. The top panel of Figure 2-4 shows the fraction of coarse particles in PM_{10}. As explained previously, geological material dominates the coarse particle fraction so coarse mass is a reasonable surrogate for PM_{10} fugitive dust contributions. The bottom panel shows the fraction of crustal material derived from elemental measurements on IMPROVE samples, adjusted for unmeasured oxides. PM_{10} and $PM_{2.5}$ mass concentrations in these plots also include secondary aerosol contributions, mostly from ammonium sulfate. The fractions therefore represent lower limits on the amounts of primary particles contributed by suspended dust.

The top panel shows coarse mass constituting 28% to 65% of PM_{10}, with typical fractions of 40% to 60% in western states and 30% to 40% in eastern states. The bottom panel shows soil constituting 3% to 30% of $PM_{2.5}$, with typical fractions of 5% to 10%. Part of the reason that the soil fraction in $PM_{2.5}$ is lower in the eastern U.S. than in the west is that secondary ammonium sulfate is a much larger fraction of $PM_{2.5}$ in the east. These background proportions are similar to those found at the mostly urban sites summarized in Table 2-3. Continental background contributions result from a variety of source emissions, some of which are not specified in Table 2-1 and several of which may originate outside the boundaries of the United States.

The major global dust producing regions are located in a broad band of arid and semi-arid lands centered on approximately 20° to 25° north latitude that extends from West Africa to northern China (Middleton et al., 1986). These areas experience a high frequency of dust storms that occur on 15 to more than 80 days per year. In the southern hemisphere, Australia also contains a significant, but less influential global dust source (McTainsh, 1986).

Figure 2-4. Fractions (percent) of PM_{10} and $PM_{2.5}$ at United States background monitors from the IMPROVE network (courtesy of J. Sisler, National Parks Service, http://alta_vista.cira.colostate.edu/).





The area in and around Africa's Sahara Desert is a major source of global dust. The Sahara Desert consists of: 1) broad alluvial plains within the Bodélé Depression in Niger and Chad; 2) southern Mauritania, northern Mali, and south central Algeria; 3) southern Morocco and western Algeria; 4) the southern fringe of the Mediterranean Sea in Libya and Egypt; and 5) the northern Sudan.

In the Middle East, the alluvial plains of southern Mesopotamia and the desert areas of Syria, Jordan, and northern Saudia Arabia are major global dust sources (Middleton, 1986; Pease et al., 1998) as are parts of Iran along the Makran coast and the Thar Desert and alluvial plains of the Amu Darya in northern Afghanistan. Important dust sources are also located in the Kara Kum Desert, parts of Kazakhstan, and in a zone extending from the southern Caspian Sea through Turkmenistan, Uzbekistan, and into Tajikistanthe (>40 dust storms/year, Pye, 1987). Major dust storms with very high sediment concentrations are also frequent in northern China with the most common occurrences in the Ordos, Takla Makam, and Gobi deserts, the Hexi and Gansu corridors, and the loess plateau region of Inner Mongolia (Middleton, 1986; Derbyshire et al., 1998).

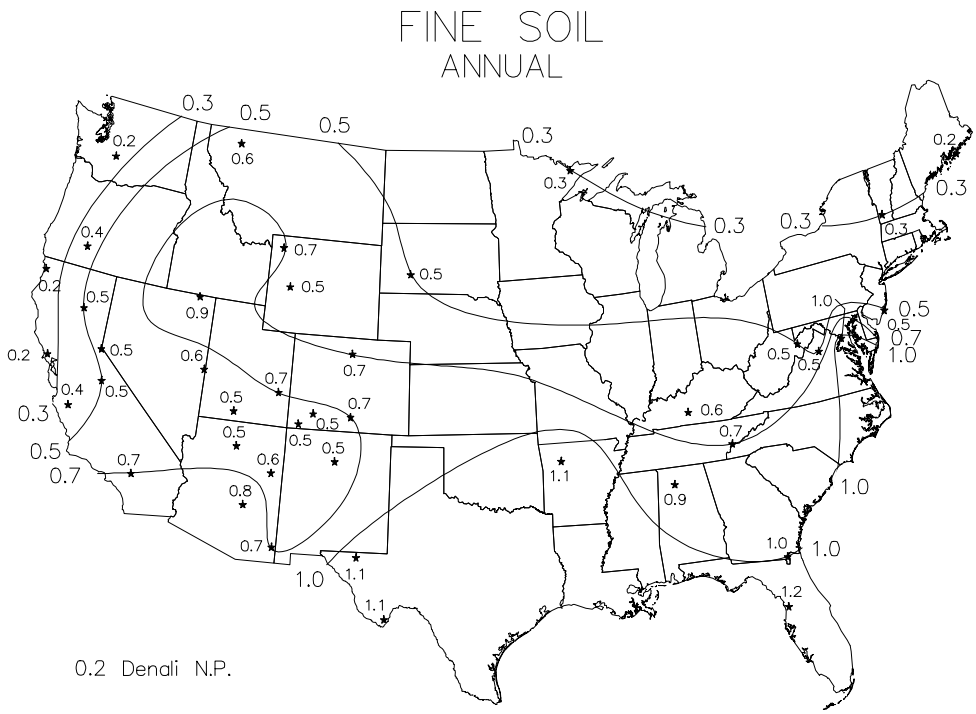
In the United States, dust storms are most frequent in the southern Great Plains area of northwest Texas and eastern New Mexico, with secondary centers in the southwestern deserts of California and Arizona (Orgill and Sehmel, 1976; Nickling and Brazel, 1984; Pye, 1987). Nickling and Brazel (1984) report that the frequency of dust storms in Arizona from 1965 to 1985 ranged from less than one per year at Tucson and Winslow to 1.6 per year at Yuma and 2.7 per year at Phoenix.

Although the major global dust sources in Africa and Asia are a considerable distance from the continental United States, they can contribute to background dust concentrations in some regions on some occasions. During the northern hemisphere summer, dust storms are frequent in the southern Sahara and the Sahel region of western Africa that extend over the eastern Atlantic. This dust laden-air reaches the Caribbean (Prospero and Nees, 1986), southern Florida (Prospero et al., 1989) and South America (Prospero et al., 1981; Swap et al., 1992) on several occasions.

Long term measurements made at Miami (Prospero and Nees, 1986) indicate that African dust is a detectable source of mineral aerosols in this region. Gatz and Prospero (1998) show an influx of African dust to the southeastern and Gulf coasts of the U.S. that traversed northeast from the Gulf of Mexico as far as central Illinois. This dust resulted in higher concentrations of Si and Al that were not consistent with local or regional dust sources. Shaw (1980) recorded transport of dust from desert sources in Asia to the Hawaiian Islands.

The wind erosion category in Table 2-1 does not represent the emissions causing these background levels. This category is dominated by windblown dust from disturbed agricultural fields and does not include the potential effects of transport from outside the United States. The fractions of fugitive dust in Table 2-3 are also positively biased because they do not subtract background dust contributions from the urban dust contributions. Figure 2-5 shows that these soil levels range from 0.2 $\mu g/m^3$ near the west coast to 1.0 $\mu g/m^3$ near

Figure 2-5. Background concentrations of $PM_{2.5}$ soil ($\mu g/m^3$) at United States background sites from the IMPROVE network (courtesy of J. Sisler, National Parks Service, http://alta_vista.cira.colostate.edu/).



the east coast and in southwest Texas. Soil concentrations in the inland western states are ~0.5 to 0.7 $\mu g/m^3$. The eastern values are based on a small number of monitoring sites, some of which are closer to urban dust sources than those in the west.

2.6 Hypothesis Testing

This section provides information on the first three hypotheses advanced in Section 1.

1. **Lack of accounting for secondary aerosol contributions in estimating fugitive dust fractions.** Subtracting secondary sulfate and nitrate contributions from PM_{10} prior to estimating the fugitive dust fraction in source apportionment studies results in a higher fractional geological contribution. There is still a large discrepancy between the proportion in emissions inventories and the fraction in ambient samples. Although $PM_{2.5}$ source apportionment studies are limited, it is likely that the discrepancy will be larger for this size fraction in which geological contributions are much lower. A more systematic examination of the fugitive dust proportion in ambient samples will be possible when data become available from EPA's $PM_{2.5}$ speciation network at representative community exposure sites throughout the United States.

2. **Lack of accounting for global and regional crustal contributions to urban concentrations.** Spatial distributions of background annual average $PM_{2.5}$ soil contributions range from 0.5 to 1 $\mu g/m^3$ in the United States. These should be subtracted from urban $PM_{2.5}$ fugitive dust contributions prior to determining proportions for comparison with local emissions inventories. Higher soil contributions to some samples in Florida and other areas may be caused by long-range transport of Sahara dust that are not accounted for in inventories. These global-scale events are not common occurrences, however, and result in a minor contribution to annual averages. More extensive IMPROVE monitoring in the eastern United States coupled with data from the $PM_{2.5}$ speciation network should allow better estimates of background contributions to urban fugitive dust levels. More specific chemical and physical markers need to be developed and measured in source and receptor samples to distinguish background from other fugitive dust sources, as well as different types of dust emitters from each other.

3. **Incompatible temporal and spatial averaging of fugitive dust emissions relative to ambient PM measurements.** Data presented here are insufficient to examine the incompatibilities between ambient and emission estimates of fugitive dust contributions. Source apportionment examples are for specific time periods and locations that do not correspond to or represent the locations and time periods of national inventories. Comparisons with location-specific inventories as part of these studies show dust emission fractions similar to those in the national inventories. Neighborhood-scale studies show that there may be large differences between dust contributions for monitors separated by no more than 5 km. A more comprehensive review of spatially dense monitoring networks designed to assess zones of influence around dust sources is needed, as are further studies that minimize differences between spatial and temporal scales of ambient measurements and inventories. Currently recognized incompatibilities do not explain the large differences between local inventories and source contribution estimates derived from receptor modeling.

3. FUGITIVE DUST EMISSIONS AND RATES

This section describes the mechanisms that cause dust suspension from unpaved roads, paved roads, construction, and wind erosion. It summarizes how these emissions are measured and related to available activity data.

3.1 Fugitive Dust Emissions Inventory Methods

Fugitive dust emission rates such as those presented in Table 2-1 are determined by the following relationship:

$$E_{jkl} = R_{jkl} \times K_{jkl} \times A_{jkl} \times (1 - P_{jkl}) \tag{3-1}$$

where

E_{jkl} = Emission rate from source type j over time period k and area l.

R_{jkl} = Rate of emissions (emission factor) for a specific size fraction per unity of activity for source type j over time period k and area l.

K_{jkl} = Particle size reduction applied to R_{jkl} when E_{jkl} is intended to represent a particle size fraction different from that represented by R_{jkl} (e.g., when $PM_{2.5}$ emissions are desired and emission factors are only available for PM_{10} or TSP). This factor is likely to be different for different source types j, time periods k, and area l.

A_{jkl} = Activity that causes dust emissions for source type j over time period l.

P_{jkl} = Fractional reduction due to emission controls applied to source j over time period k and area l.

Each of the components of E_{jkl} is empirically derived from a limited number of tests. These tests are intended to represent the entire population of emission factors, activity levels, size distributions, and emission reduction effectiveness. Averaging periods are typically for a year or season and averaging areas are typically the sizes of counties or states. As noted in Section 2, each of these components of fugitive dust emission rate contains uncertainties when applied to a specific situation.

3.2 Processes that Affect Dust Suspension

Fugitive dust emissions depend on particle sizes, surface loadings, surface conditions, wind speeds, atmospheric and surface moisture, and dust-suspending activities. Emission rates and control measures are also closely related to these properties.

3.2.1 Particle Sizes

The "silt" fraction of surface dust is most often used as a surrogate for suspendable particles. Silt consists of particles with a geometric diameter <75 μm as determined by sieving dried soil samples acquired from surface loading tests. The 75 μm geometric diameter corresponds to an aerodynamic diameter of ~120 μm because the aerodynamic diameter varies inversely with the square root of the density (Hinds, 1999), which is ~2.65 g/cm^3 for minerals. Similarly, a 10 μm aerodynamic diameter dust particle has a ~6 μm geometric diameter and a 2.5 μm aerodynamic diameter dust particle has a geometric diameter of ~1.5 μm. Little is known about the PM_{10} and $PM_{2.5}$ in surface dust as these fractions are too small to be determined by simple sieving methods.

Particle size distributions have been determined by sieving samples from different types of soils and recording these in soil surveys. These surveys have been used for agriculture and construction/engineering purposes since the early 1900s in many parts of the United States and are commonly available at county agricultural extension offices. The particle sizing procedure (American Society for Testing and Materials, 1997, 1998) most commonly followed for soil surveys creates a soil/water suspension in which soil aggregates are broken into their component parts prior to sieving. While the particle size distribution of the disaggregated sediment is useful for agricultural, construction, and other land uses, it is not entirely applicable for estimating air pollution emissions. This sieving method does not estimate the size of the dust aggregates that are entrained and suspended by surface winds or human activities. The silt fraction is determined by dividing the weight of material passing through the 75 μm sieve by the weight of material presented to a stack of sieves.

Gillette et al. (1980) applied two methods to determine the particle and aggregate sizes in soil that might be entrained by winds. The first method ("gentle sieve") consists of drying a soil sample and sieving it gently with about twenty circular gyrations parallel to the plane of the sieve. The second method ("hard sieve") consists of up to one-half hour of vigorous shaking (usually using a shaking machine). Cowherd et al. (1990) used a rotary sieving procedure described by Chepil (1952) to estimate the modal aggregate size of sediment samples removed from unpaved roads. The gentle sieve method is assumed, without quantitative validation, to be a more suitable approach for determining potential wind erosion properties of bare soil because it attempts to sample the sediment with its *in situ* characteristics intact. Silt fractions and amounts determined by the hard sieve method probably provide a reasonable indicator of small particles from roads where vehicle tires abrade the surface. Threshold suspension velocities (Gillette et al., 1980) for windblown dust apply to soil characteristics obtained by the gentle sieve method. Appendix C-2 of AP-42 (U.S. EPA, 1999) provides detailed procedures for the hard sieve method applicable to other emissions.

The size distribution of dust particles affects the suspension process. A flat bed of particles with diameters <20 μm is difficult to suspend by wind. Bagnold (1937) showed that fine Portland cement could not be entrained by wind velocities in excess of 1 m/s at the surface. In this situation, there is no large cross section for wind to act on. In addition, adhesive forces such as van der Waals, electrostatic, and surface tension of adsorbed liquid films (Hinds, 1999) increase the force required to entrain the particles.

Suspension of PM_{10} and $PM_{2.5}$ is reduced by larger non-erodible particles. Particles with diameters exceeding ~800 μm shelter smaller particles in their lee (Chepil, 1942). Gillette and Stockton (1989) sprinkled glass spheres with diameters ranging from 2,400 to 11,200 μm onto a bed of glass spheres with diameters of 107 to 575 μm and found major reductions in the horizontal flux of the smaller particles. However, Logie (1982) found that erosion of a sand surface was enhanced when low concentrations of larger non-erodible obstructions were present on the surface. Logie (1982) hypothesized that the increased erosion was due to wind acceleration around the isolated obstructions that scoured the loose sand. Bagnold (1941) estimated that 800 μm particles can be separated from surfaces under high winds, although their large masses cause them to rapidly settle to the surface. Rosbury and Zimmer (1983) and Flocchini et al. (1994) observed higher dust emissions from unpaved road surfaces with lower silt contents and higher gravel content. When acted on by vehicle tires, larger gravel particles replenished the quantity of smaller suspendable particles by enhancing surface abrasion.

Carvacho et al (1996) have developed a method that directly measures the reservoir of PM_{10} available for suspension in a bulk soil sample. Pulses of air are run through a fluidized bed containing the sample and the suspended dust is sampled through a size-selective inlet onto a series of parallel filters. After a few pulses, one of the filters stops sampling. After a few more pulses, another filter stops sampling and so on. Each filter is separately weighed and the collected mass is plotted as a function of the aggregate agitation (as determined by the number of pulses). A maximum mass loading is reached after many pulses when the suspendable PM_{10} is depleted. This maximum varies between different types of dusts much more than it varies for repeated samples of the same dust. Campbell and Shimp (1998) have related this PM_{10} suspension potential to the silt measurements in California soil surveys to improve their PM_{10} emission estimates.

Silt fractions or quantities appear as explicit variables in many of the emission factors cited below. The processes related to particle size indicate that actual emissions of PM_{10} and $PM_{2.5}$ are influenced by more detailed size distributions above and below the 75 μm geometric diameter that specifies silt content.

3.2.2 Surface Loading

Most soil surfaces are limited reservoirs; suspendable dust is depleted after a short time in the absence of direct abrasion. This depletion is represented as a negative exponential (Anspaugh et al., 1975; Linsley, 1978) or inverse (Garland, 1983; Reeks et al., 1985; Nicholson, 1993) function of time.

As noted above, depletion of fine particles often results in the exposure of larger non-erodible sediments that shield the suspendable particles from the wind. The larger non-erodible elements also absorb momentum, thereby decreasing the wind's ability to erode the surface (Marshall, 1971; Raupach, 1992). When surfaces are continually disturbed by very intense winds, by vehicular movement, or by other human activities, unlimited reservoirs are created that emit dust whenever winds exceed threshold suspension velocities. Suspendable dust loadings may vary substantially, even over periods of a few minutes, when there are no mechanisms to replenish the reservoir.

Surface loadings are determined by sweeping or vacuuming loose particles from several specified areas within a fugitive dust source. These samples are dried and weighed, then divided by the area sampled to determine the mass of dust per unit area. Appendix C-1 of AP-42 (U.S. EPA, 1999) provides detailed procedures. The same samples are usually sieved to determine the silt fraction and the quantity of larger particle sizes. Surface loadings vary substantially with space and time. Zimmer et al. (1992) measured silt loadings that varied by a factor of twenty at different times for the same paved road in Denver, CO. South Coast Air Quality Management District (1991) reported paved road silt loadings from 0.112 to 1.83 g/m^2 for different areas with similar road and traffic conditions in the Los Angeles area.

Kuhns and Etymezian (1999) mapped silt loadings on streets in Las Vegas, NV, using light scattering photometers located in a vehicle wheel well to measure dust suspended by the tires and on the hood to measure ambient levels. After adjusting for vehicle speed, they found a consistent relationship between silt loadings determined by sieving and those determined by one-second averages of light scattering near the road surface. Surface silt loadings on highly traveled streets were much lower than those on less-heavily used residential side streets. Visible trackout from construction sites yielded orders of magnitude higher silt loadings than loadings on heavily traveled roads.

3.2.3 Surface Conditions

Surface conditions refer to the landform shape and cohesion of a potential dust reservoir. The effects of landform shape on dust suspension are embodied in the concept of "surface roughness." Surface roughness is related to the heights of obstructions within and around exposed dust areas. Agriculturists often orient their furrows perpendicular to prevailing winds, or plant rows of trees upwind of their fields, to minimize soil losses from wind erosion by increasing surface roughness. Larger surface roughness decreases the force exerted by the wind on suspendable surface particles, thereby decreasing emissions. However, larger surface roughness increases vertical turbulence that can mix suspended particles higher into the atmosphere for longer transport distances.

The aerodynamic roughness length is the apparent distance above the surface at which the average wind speed approaches zero. In reality, wind speed does not become zero at this level, but it deviates from the logarithmic increase of wind speed with height that is commonly found in the atmosphere. Aerodynamic surface roughness is ~3% to ~12% of the height of obstructions in and around an exposed area (Greeley and Iversen, 1985); it is site-specific and quantified with wind speed measurements taken at different elevations between ~1 m and ~10 m above ground level. The ratio of wind speeds at lower levels is plotted against the logarithm of the measurement height and extrapolated to a wind velocity of zero. The intersection with the elevation axis at wind speed equals zero is the surface roughness. In practice, estimates from many hours of wind measurements are averaged to determine typical surface roughness. The slope of this relationship is termed the "friction velocity" and indicates the wind shear forces near an erodible surface (Pasquill and Smith, 1983). The presence of large particles or obstructions that increase surface roughness attenuates wind erosion by absorbing a significant fraction of the downward momentum flux from the air

flow above (Wilson et al., 1998). While the principles are known, the effects of changes in surface roughness on fugitive dust emissions are not well quantified.

Surface roughness may be associated with many degrees of surface cohesion. Natural weathered soil in arid environments tends to develop a crust over time that is highly resistant to suspension (Belnap and Gillette, 1997). This surface is easily broken by footsteps, let alone by vehicular traffic and livestock grazing (Gillette et al., 1979; 1982). A large reservoir of suspendable material is often available beneath this crust. Desert pavements require many decades to reform, as evidenced by still-visible wagon tracks along emigrant trails blazed during the 1840s and 1850s in the western U.S.

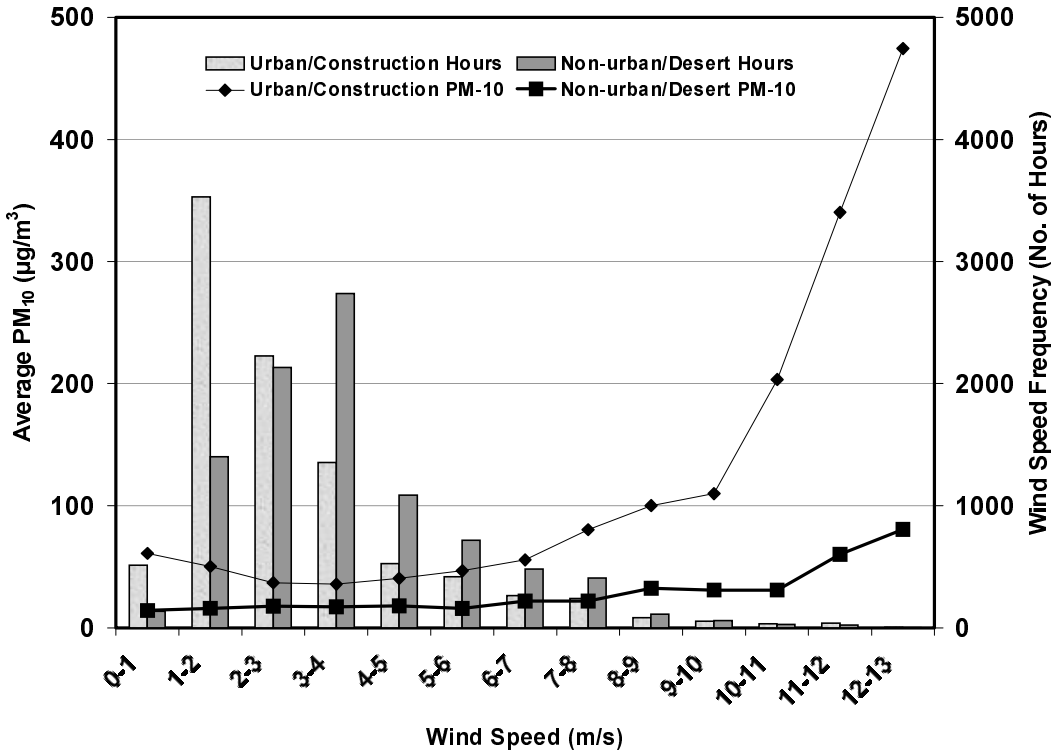
Gillies et al (1999) estimated the strength and resilience of unpaved road surfaces by measuring the vertical force (kg/cm^2) needed for penetration with a Proctor Penetrometer®. This is similar in concept to the "rupture moduli" applied by Gillette et al. (1982). Force measurements were taken across the road width every 0.25 m on different occasions and for different dust suppression treatments. Hard surfaces, similar to but stronger than natural desert crusts, experienced a brittle failure after application of a large force. The surface shattered, creating small aggregates and holes. Loose particles were exposed, as were edges of the surface that could be further ground down by tire wear. A flexible surface created by a polymer emulsion was penetrated at a relatively low force, but the size of the penetration was limited to the diameter of the Penetrometer and no aggregates were created. While surface strength may be a good indicator of emission potential for brittle surfaces, the nature of penetration is just as important. Vegetated and moist surfaces exhibit elastic characteristics similar to those of the polymer emulsion.

3.2.4 Wind Speeds

High wind speeds provide the energy needed to suspend loose particles from a surface; turbulence associated with these winds elevates particles to high altitudes where they can transport over long distances (Prospero et al., 1970, 1981, 1989; Gillette and Blifford, 1971; Gillette et al., 1972; 1978a; 1978b; Prospero and Carlson, 1972; Duce et al. 1980; Gillette, 1980a, 1980b, 1981). Wind erosion occurs in urban as well as non-urban areas. Figure 3-1 compares PM_{10} concentrations averaged for different wind speeds near a construction site in Las Vegas, NV and at a location in the nearby non-urban desert. The urban site shows higher concentrations at wind speeds of 0 to 2 m/s owing to accumulation of nearby emissions under stagnant conditions. PM_{10} levels begin to increase at wind speeds of 4 to 5 m/s, attributable to windblown dust emissions from nearby land parcels denuded for new construction (Chow et al., 1999; Chow and Watson, 1997b). Large increments in PM_{10} are not seen, however, until wind speeds exceed 7 m/s with concentrations increasing rapidly for wind speeds in excess of 10 m/s. Fortunately, these high winds are infrequent, occurring for only 83 hours during 1995. Several of these hours were consecutive, however, and were associated with 24-hour average PM_{10} exceeding 150 $\mu g/m^3$.

The non-urban desert site experienced a similar frequency of high winds, but it does not show significant increases in emissions until these speeds exceed 11 m/s. A slight increase in PM_{10} is noticeable for wind speeds above 8 m/s. This example refutes the

Figure 3-1.. Average PM_{10} classifed by wind speed from hourly beta attenuation monitor (BAM) measurements at an Urban/Construction site and a Non-Urban/Desert site near Las Vegas, NV during 1995 (Chow and Watson, 1997b; Chow et al., 1999). Wind speeds were measured at 10 m above ground level.



argument that most urban dust derives from natural surfaces. Only when wind speeds are high and persistent do undisturbed areas provide measurable, though not major, fractions of PM_{10} or $PM_{2.5}$ in the atmosphere. Non-urban surfaces disturbed by off-road traffic, agricultural operations, and construction should not be classified as natural sources because they were created by human intervention.

Chepil and Woodruff (1963) and Gillette and Hanson (1989) show that the amount of dust suspended by wind depends on particle size distributions, wind speed at the surface, surface roughness, relative amounts of erodible (<2 mm diameter) and non-erodible (>2 mm diameter) material, and the cohesion of the soil particles with one another. Values for each of these variables affect other variables. For example, a higher moisture content increases cohesion among particles and shifts the size distribution to larger particles. Larger agglomerations of small particles increase surface roughness, thereby decreasing wind speeds at the surface.

The effects of all of these variables are embodied in a threshold friction velocity that is experimentally determined by placing a wind tunnel over an example of the affected soil

and measuring the surface speed at which soil movement first becomes visible (Zingg, 1951; Gillette, 1978a, 1978b, 1980, 1981; Raupauch et al., 1980; Borrmann and Jaenicke, 1987; Rajendran and Fraftz, 1992; Visser, 1992; Nicholson, 1993; Braaten, 1994; Giess et al., 1994). With the more common use of continuous particulate monitors (Watson et al., 1998b), threshold friction velocities might be inferred from hourly PM_{10}, $PM_{2.5}$ and wind speed measurements at ambient sampling sites as demonstrated in Figure 3-1. Averaging times of one to five minutes would provide more precise estimates than the hourly averages used in Figure 3-1.

Gillette (1980) shows threshold friction velocities that vary from 0.19 to 1.82 m/s for soils with different degrees of disturbance. Most ambient wind speed measurements such as those in Figure 3-1 are made at elevations between 5 and 10 m above ground level, and these must be translated to surface friction velocities to determine suspension. This is done using estimates of surface roughness and friction velocities from the actual or similar sites (U.S. EPA, 1999). For this range of surface threshold values, emissions will be initiated at ambient wind speeds (measured at 7 m above the ground level, the height of most National Weather Service wind sensors) between 7 and 10 m/s (26 to 37 km/h). Even though emissions begin at these velocities, the wind force contains insufficient energy to suspend very much of the erodible soil mass. The amount of dust suspended increases at approximately the cube of the wind speed above the threshold velocity. This is consistent with the measurements in Figure 3-1.

Particles suspended into the atmosphere are acted upon by gravity in a downward direction and by atmospheric resistance in an upward direction. Every particle attains an equilibrium between these forces at its terminal settling velocity. The settling velocity increases as the square of the particle diameter, and linearly with particle density (Friedlander, 1999). For very small particles (<10 μm diameter), turbulent air movements in wind storms can counteract the gravitational settling velocity and such particles can remain suspended for long times (Sehmel, 1980, 1984; Slinn, 1982). Transport distance depends on the initial elevation of a particle above ground level, the horizontal wind velocity component at the particle elevation, and the gravitational settling velocity.

Pye (1987) shows vertical profiles for different sized particles that might be elevated through a 100 m depth during a wind storm. The particles smaller than 10 μm are nearly uniformly distributed through this depth, while the larger particles exhibit much higher concentrations closer to the surface. Terminal settling velocities dominate most situations for >10 μm particles in the absence of violent winds.

3.2.5 Moisture Content

Water adhering to soil particles increases their mass and surface tension forces, thereby decreasing suspension and transport. Cohesion of wetted particles often persists after the water has evaporated due to the formation of aggregates and surface crusts. Soil moisture content is determined from representative samples swept or vacuumed from the potentially emitting surface and stored in an airtight container. A portion of the sample is weighed before and after 24-hour heating at ~110 °F in a laboratory drying oven, with the difference being equal to the evaporated water (Weems, 1991; Ley, 1994). Low temperatures are used

to minimize volatilization of materials other than water that might affect the weight change. Appendix C-2 of AP-42 (U.S. EPA, 1999) provides detailed procedures.

Rosbury and Zimmer (1983) found that moisture content affects the ejection of particles by vehicles, as well as the strength of the road bed and hence its ability to deform under vehicle weight. The addition of water to create surface moisture contents exceeding 2% resulted in >80% reductions for PM_{10} emissions compared to a control surface with an average moisture content of 0.56% (Flocchini et al., 1994). Road surface-moisture content enhances the strength of surface crusts and the stability of aggregates (Bradford and Grosman, 1982; Lehrsch and Jolley, 1992).

Kinsey and Cowherd (1992) show how watering reduces emissions at a construction site. Significant dust control benefits are derived initially by doubling the area that is watered; however, benefits are reduced as more water is applied to the site. Ultimately, control efficiency is limited because grading operations are continually exposing dry earth and burying the moistened topsoil.

Excessive moisture causes dust to adhere to vehicle surfaces so that it can be carried out of unpaved roads, parking lots, and staging areas. Carryout also occurs when trucks exit heavily watered construction sites (Englehart and Kinsey, 1983). This dust is deposited on paved (or unpaved) roadway surfaces as it dries, where it is available for suspension far from its point of origin. Fugitive dust emissions from paved roads are often higher after rainstorms in areas where unpaved accesses are abundant, even though the rain may have flushed existing dust from the paved streets.

The same amount of moisture affects different dust surfaces in different ways. Moisture capacities of different geological materials are documented in soil surveys. Soil surveys include several indicators of the ability of soils to absorb moisture, with the most common being the "plastic limit" and "liquid limit". The plastic limit is the moisture content at the threshold between the plastic and semi-solid states of a clayey soil. It is determined by repeatedly rolling an ellipsoidal pellet of soil on a ground glass plate until a slender thread of soil forms and then crumbles. The liquid limit is the moisture content at the threshold between the plastic and semi-liquid states of a clayey soil. It is determined using a standard cup of soil, grooving tool, and tapping device to determine the moisture content when a groove carved in the soil closes (Das, 1998). Soil surveys also report the infiltration rate (the movement of water through soil layers) and field moisture capacity.

The actual moisture content at a given time or place is not recorded and must be estimated. Thornthwaite (1931) proposed the ratio of precipitation to evaporation as an indicator of the availability of moisture for soils. Thornthwaite's major concern was the agricultural potential of land in different areas. The precipitation-evaporation effectiveness index (P-E index) is ten times the sum of the monthly precipitation to evaporation ratios. Thornthwaite (1931) classified North American regions as wet (P-E index > 128), humid (64 < P-E index < 128), sub-humid (32 < P-E index < 64), semi-arid (16 < P-E index < 32), or arid (P-E index < 16). Much of the western U.S. is in the arid and semi-arid categories. The P-E index has been used to estimate the moisture content of different soils with emission

factors for different surface types. Cowherd et al. (1988) provide a map of average evaporation rates for the 1946-1955.

Precipitation events and the P-E index are crude methods of estimating soil moisture, which is likely to be affected by snow, fogs, and high humidity, and to decrease over several days following heavy precipitation. The moisture content of soils varies throughout the year depending on the frequency and intensity of precipitation events, irrigation, and relative humidity and temperature of the surrounding air. Large amounts of rain falling during one month of a year will not be as effective in stabilizing dust as the same amount of rain interspersed at intervals throughout the year. Vehicle traffic enhances moisture evaporation by increasing air movement among the surface particles and exposing dry soil below the moist surface. Trees and other natural or manmade formations that evapo-transpire or cast shadows can also enhance or retain soil moisture content. A more realistic model might use hourly rainfall, snowfall, relative humidity, and traffic volumes with monthly average Class A Pan evaporation rates to estimate soil moisture content.

3.2.6 Vehicular Movement

The most common urban dust-suspending activity is vehicular movement on paved roads, unpaved roads, parking lots, and construction sites. Vehicle shape, speed, weight, number of wheels as well as previous history (e.g., dust acquisition for trackout) interact with different road surfaces to change the particle size, surface loading, wind effects, and surface moisture. Vehicular traffic adds to particle suspension because tire contact creates a shearing force with the road that lifts particles into the air (Nicholson et al., 1989). Moving vehicles also create turbulent wakes that act much like natural winds to raise particles (Moosmüller et al., 1998). Natural crusts are often disturbed by vehicular movement, increasing the reservoir available for wind erosion.

Dust on paved roads must be continually replenished; minimizing the deposition of fresh dust onto these surfaces is a viable method for reducing their PM emissions. Dust loadings on a paved road surface build up by being tracked out from unpaved areas such as construction sites, unpaved roads, parking lots, and shoulders; by spills from trucks carrying dirt and other particulate materials; by transport of dirt collected on vehicle undercarriages; by wear of vehicle components such as tires, brakes, clutches, and exhaust system components; by wear of the pavement surface; by deposition of suspended particles from many emission sources; and by water and wind erosion from adjacent areas (Chow and Watson, 1992; Chow et al., 1990).

The relative contribution from each of these sources is unknown. Axetell and Zell (1977) estimated typical deposition rates of 67.8 kg/km over a 24-hour period for particles of all sizes from the following sources: 1) 42% from mud and dirt carryout; 2) 17% from litter; 3) 8% from biological debris; 4) 8% from ice control compounds (in areas with cold winters); 5) 8% from erosion of shoulders and adjacent areas; 6) 7% from motor vehicles; 7) 4% from atmospheric dustfall; 8) 4% from pavement wear; and 9) less than 1% from spills. Axetell and Zell (1977) cite these fractions without describing the methodology used to estimate them.

Unpaved roads and other unpaved areas with vehicular activity are unlimited reservoirs when vehicles are moving. When regularly traveled, these surfaces are always being disturbed and wind erosion seldom has an opportunity to decrease surface loadings or increase the surface roughness sufficiently to attenuate particle suspension. The grinding of particles by tires against the road surface shifts the size distribution toward smaller particles, especially those in the PM_{10} fraction. Pinnick et al. (1985) found the distribution of particle sizes within a vehicle-created dust plume to be bimodal, with one mode peaking at ~50 μm another mode peaking at ~2.5 μm. Patterson and Gillette (1977) reported a similar distribution for naturally generated windblown dust plumes, with fewer large particles in the natural plume than in the vehicle-generated plume. The bimodal distribution was attributed to grinding processes caused by tires for the vehicle dust (Pinnick et al., 1985) and to a sandblasting process for wind-generated dust (Patterson and Gillette, 1977). Nicholson et al. (1989) show particle size and emission rate increasing with vehicle velocity, consistent with higher energy transfer through surface contact and turbulent wakes at higher speeds.

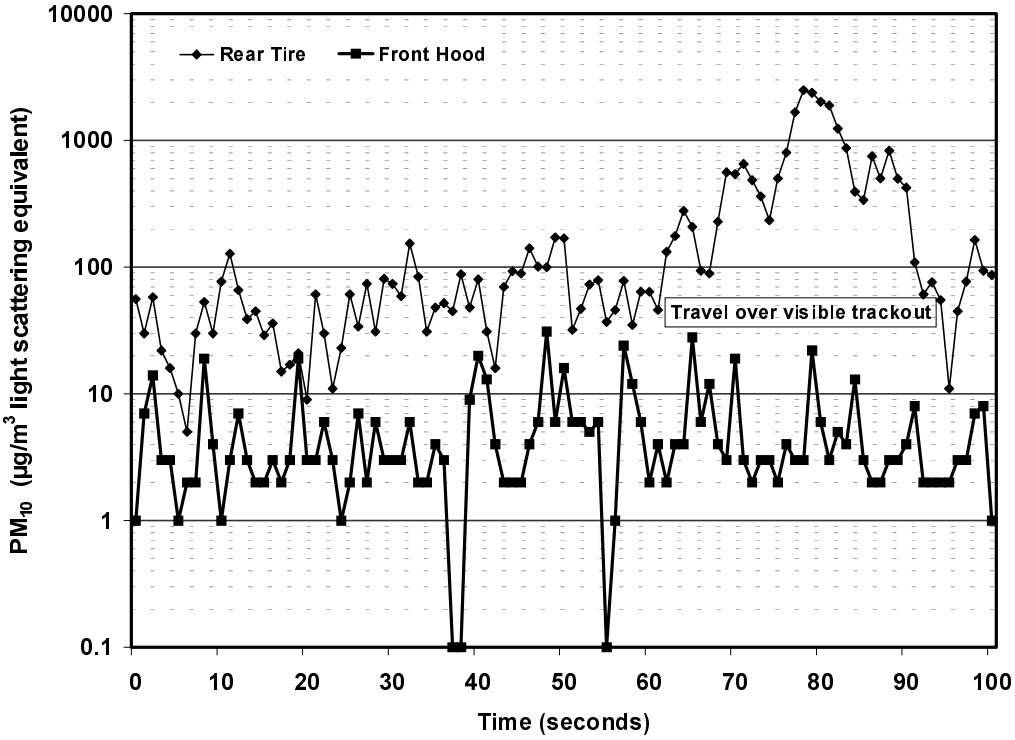
Dyck and Stukel (1976) hypothesized that vehicle weight and road type influence dust emissions. Mollinger et al. (1993) found the shape of vehicles to have a large impact on the amount of dust suspension; a cylinder, an elliptical solid, and a rectangular solid were mounted on a pendulum that swung back and forth over dust-covered test areas. After twenty passes by the cylinder and elliptical solid, 65% and 45% of the dust remained in the test area, respectively. After twenty passes by the rectangular solid traveling at the same velocity, less than 20% of the dust remained. Moosmüller et al. (1998) found that only high profile vehicles, such as semi tractor trailers, produced sufficient turbulence to suspend dust along an unpaved shoulder next to a paved road. Normal passenger vehicles and pickup trucks produced negligible shoulder emissions when traveling at speeds of 80 to 96 km/h.

Figure 3-2 shows the effect of several variables, especially surface loading, on emissions from a paved roadway measured by the particle scattering methods of Kuhns and Etymezian (1999). PM_{10} (light scattering equivalent) measured behind a vehicle tire on a clean-looking roadway was 10 to 100 times the PM_{10} in the surrounding air. PM_{10} measured behind the tire increased by another order of magnitude when the vehicle passed over a visible deposit from construction site trackout. Figure 3-2 demonstrates the variability of road emissions over short distances. It also suggests a method to evaluate roadway emissions potential that would couple high time response PM_{10} measurements with Geographic Positioning System (GPS) tracking. These continuous measurements may be more closely related to actual emissions than sporadic silt content and surface loading samples.

3.2.7 Industrial Processes and Construction

In addition to paved and unpaved roads and disturbed areas, many industrial and construction sites that use mineral products have storage pile, material conveyance and loading, digging, dozing, grading, scraping, and blasting activities. These activities create dust reservoirs, mechanically inject dust into the atmosphere, and present targets for wind erosion. Most of these operations are affected by the same variables described above, but the frequency, spatial extent, and magnitudes differ from those of urban and non-urban dust

Figure 3-2. PM_{10} equivalent concentrations on a paved road in Las Vegas, NV on the front hood (~1 m above ground level [agl]) and behind the right front tire (~0.25 m agl) of a vehicle moving at 10 mph. Concentrations behind the tire increased considerably with travel over a surface with visible trackout from a construction site. One-second PM_{10} concentrations are estimated using a DUSTRAK® photometer calibrated with Arizona road dust.



sources. Since most industrial operations are confined to private property, the combined PM_{10} and $PM_{2.5}$ emissions that travel beyond the property limits are of greatest concern to NAAQS attainment. Visible clouds and deposition of large particles (>30 µm) on neighboring communities are often nuisances that need to be addressed.

Most modern industrial sites have designed the placement of their operations to minimize both emissions and transport of dust outside of property boundaries. Large blending domes, for example, are used in the cement industry to contain dust and to obtain a more homogeneous composition of the feed material to the kiln, thereby making better use of raw materials and minimizing upsets. Underground transport systems and covered conveyers are commonly used to minimize product loss. A fortunate by-product of these technological advances is to reduce fugitive dust from materials handling and plant upsets while increasing product volume and quality.

3.3 Dust Control Measures

The fractional reduction owing to fugitive dust controls (P_{jkl} in Equation 3-1) is variable and uncertain. Dust controls are applied with various degrees of effectiveness and diligence, often in response to local nuisance complaints rather than as part of a comprehensive emission reduction strategy. Mitigation methods include some combination of reducing suspendable dust, preventing its deposit, stabilizing it, enclosing it, and reducing the activities that suspend it. Table 3-1 identifies several control methods and the sources to which they apply.

Emission reduction plans usually assume that some sources or facilities will not be fully in compliance with regulations. For industrial sources, permits are issued, the processes and emissions are fairly well known, and enforcement by regulatory agencies is ongoing. For fugitive dust, however, controls and their enforcement differ substantially from region to region. Reasons for differences include: 1) variable air quality permitting processes for construction projects, unpaved roads, disturbed soil surfaces, etc.; 2) the temporary nature of activities such as construction projects; 3) the large number of small construction projects that can be active at a given time; 4) the intermittent nature of control actions, such as watering or use of chemical stabilizers, as opposed to continuously operating controls, such as baghouses or scrubbers; and 5) the ever changing number of contractors, material haulers, and others, many from out-of-area or out-of-state, who lack knowledge of local fugitive dust control regulations.

The success of fugitive dust control programs depends on outreach and education programs for contractors and public works agencies as well as continuing enforcement of air quality rules. Penalties must be sufficient to motivate offenders to obey the law, and not just a "cost of doing business."

3.3.1 Surface Watering

Surface watering is often applied on disturbed land at construction sites and other unpaved surfaces to reduce particle suspension by vehicles. Flocchini et al. (1994) found that the addition of sufficient water to increase the surface moisture content from 0.56% to 2% can achieve greater than 86% reduction in PM_{10} emissions. Kinsey and Cowherd (1992) found immediate dust reductions at construction sites as a result of surface watering; however, the effectiveness of this measure did not increase as more water was applied to the site. Excessive moisture content on unpaved roads can have negative effects, such as: 1) reducing the strength of the road bed; 2) enhancing road bed deformation under vehicle loading; and 3) increasing the potential for brittle failure that produces small particles to be further crushed by tires (Rosbury and Zimmer, 1983).

Excessive moisture at construction sites can increase track-out and carry-out onto paved roads when wet dust adheres to haul trucks. Track-out refers to the sediment that is attached to vehicle tires and subsequently transferred to the road surface. Carry-out is the sediment attached to the vehicle that may eventually fall off and become resuspended by other vehicles (Axetell and Zell, 1977; Brookman, 1983). Washing vehicles as they depart

Table 3-1. Urban fugitive dust emission control methods.

Control Method	Sources Controlled	Description
Street sweeping	Paved roads	Sweepers use mechanical brushes, vacuum suction, regenerative air suction, or blow-air/suction recirculation to remove street debris, litter, and dirt.
Water flushing	Paved roads	Pressurized water sprays or water with surfactants added dislodge road dust and transport it to a drain system.
Resurfacing	Paved roads	Repaving with non-erodible materials minimizes pavement cracks that trap and accumulate dust and reduces pavement abrasion.
Wet suppression	Unpaved roads, construction	Water applied to loose soils agglomerates small dust particles into larger entities that adhere to the surface, resist suspension, or deposit rapidly after suspension.
Windscreens and greenbelts	Unpaved roads, disturbed land, construction	Fabric or wooden barriers, trees, or shrubbery are placed upwind to reduce surface wind speeds or downwind to allow for horizontal impaction of suspended particles.
Traffic controls	Paved roads, unpaved roads	Lower vehicle speeds, limited road usage, restriction of heavy-duty vehicle traffic, and provision of parking and public transit opportunities reduces activity on roads that produce dust.
Carryout and trackout reduction	Construction	Wheel and truck washing on leaving a dusty site limits distribution of suspendable particles onto paved roads. A paved or stabilized area with a wheel vibrator that is constantly swept prior to exiting a job site allows residual material to fall off of vehicles.
Enclosures	Storage piles, open haul trucks	Covering piles with tarps or building structures around them limits wind erosion.
Chemical stabilization	Unpaved roads, construction sites, storage piles	Chemical agents bind particles into larger aggregates that reduce the reservoir of suspendable particles.
Vegetative stabilization	Inactive construction sites, unpaved shoulders	Ground cover and shrubbery reduces wind velocity at the surface and binds surface soil particles.
Foggers	Construction vehicles, material transfer, storage piles	Small water droplets, sometimes electrically charged, agglomerate suspended particles, thereby increasing particle size and deposition velocity.

from work areas can minimize dust track-out and carry-out. Axetell and Zell (1977) found that TSP concentrations (~84 $\mu g/m^3$) increased by 40 to 60 $\mu g/m^3$ due to uncontrolled mud track-out. Immediate cleanup with shovel and broom reduced TSP by 10 to 20 $\mu g/m^3$, and daily cleanup reduced TSP by about 5 to 10 $\mu g/m^3$. These values correspond to control efficiencies of about 30% for immediate cleanup and 15% for daily cleanup. Besides wheel washing, other cleaning procedures include street sweeping, manual broom and shovel cleanup, and water flushing to remove mud and dirt from paved surfaces after it has been tracked out. .

3.3.2 Chemical Suppression

The application of chemical suppressants on unpaved surfaces can reduce fugitive dust emissions (Cuscino et al., 1983a; Drehmel et al., 1982; Gillies et al., 1999). Watson et al. (1996b) enumerate commercially available dust suppressants. These products are classified into seven categories according to their chemical composition and the suppressant mechanism they employ:

- **Salts:** Hygroscopic compounds such as magnesium chloride or calcium chloride. They adsorb water as ambient relative humidity exceeds 50%. Water improves the adherence of the soil particles to each other, thereby reducing dust emissions. Since salts are water soluble, precipitation tends to wash them away.
- **Resin or petroleum emulsions**: Non-water-soluble organic carbon compounds suspended in water. When these emulsions are sprayed onto soil, they stick the soil particles together, and eventually harden to form a solid mass. Several emulsion products are based on tree resin, petroleum, or asphalt compounds.
- **Polymers:** Long-chain molecular compounds that act as adhesives to bond soil particles together. In theory, polymers may be able to stick to more particles than ordinary resins, or they may bridge larger particle-to-particle gaps.
- **Surfactants**: Chemicals that reduce water surface tension and allow available moisture to more effectively wet the particles and aggregates in the surface layer.
- **Bitumens**: Materials such as asphalt or road oil that act as adhesives to bond soil particles together.
- **Adhesives:** Paper mill byproducts, such as lignin sulfonate (a syrupy wood product), which form a sticky but water-soluble layer on unpaved surfaces.
- **Solid Materials:** Petroleum industry byproducts made by mixing recycled materials with soil.

Most suppressants require repeated application at frequencies on the order of weeks or months. The effectiveness of chemical suppressants depends on road surface conditions, soil composition, application intensity, traffic volume, vehicle weight, and environmental factors such as precipitation and temperature. Prior to suppressant application, the road surface often needs to be graded or wetted. Most products can be dispensed as liquids by a truck equipped with a tank and spray bar. The spraying process injects the suppressant into the road material. Solid materials can be spread and mixed into the soil or road bed with a grader.

Several studies have examined the effectiveness of different chemical suppressants on various types of unpaved roads (e.g., Cuscino et al., 1983a). The control effectiveness is used as a measure of suppressant effectiveness between untreated and treated surfaces. Rosbury and Zimmer (1983) found considerable variability in dust emission rates for untreated and chemical-suppressant-treated surfaces. Much of this variation was attributed to the effects of ambient meteorological conditions (especially precipitation), the types of vehicles traveling on the road, and the initial road conditions. Control effectiveness varied from 7% to 54% for well-mixed suppressant applications, and from insignificant to 80% for topical suppressant applications. To achieve 50% control effectiveness, monthly surface watering was needed.

Muleski and Cowherd (1987) evaluated the effectiveness of chemical suppressants on unpaved roads used by the iron and steel industry. Particles in three size fractions (<15 μm, <10 μm, and <2.5 μm) were acquired at heights up to 6 m above the surface to assess the suppressant efficiencies. Characteristics of unpaved road surfaces (such as percent silt and moisture content and the amount of loose surface material) were measured. Muleski and Cowherd (1987) report unpaved road emission rates similar to those found by Rosbury and Zimmer (1983). Average control effectiveness of ~50% or more was found within the first 30 days of suppressant application; suppressant effectiveness was negligible 30 days after suppressant application.

Grau (1993) tested suppressant-treated soil specimens under controlled laboratory conditions. Wind erosion (one-minute blasts from 80 km/h and 160 km/h air jets at 20° from the horizontal), rainfall, and jet fuel spills were simulated. Forty-nine suppressants were evaluated by visual observations of entrainment or lack thereof. Of these, eleven suppressants were deemed to be effective in laboratory simulations.

Flocchini et al. (1994) estimated PM_{10} fugitive dust emissions from non-urban unpaved roads and evaluated the effectiveness of chemical suppressants in California's San Joaquin Valley. The effectiveness of surface watering, gravel cover, lignin sulfonate, magnesium chloride, oiling, and nonhazardous crude oil were assessed by measuring PM_{10} and size-segregated particle mass concentrations at upwind and downwind locations. Vertical concentration gradients for particles above and below $PM_{2.5}$ were acquired (Cahill et al., 1990). Mass concentrations at 10 m downwind and 3.3 m in height were adjusted for background concentrations to assess the effect of vehicles traveling on the unpaved roads. Surface characteristics, soil type, surface loading per unit area, moisture content, and silt content (< 75 μm particle diameter) were measured. Surface samples were resuspended in the laboratory to determine their relative potentials for PM_{10} emissions. Surface watering and road oiling reduced PM_{10} emissions by 87 ± 6% and 59 ± 12%, respectively. Reducing vehicle travel speeds on unpaved roads from 40 km/h to 16 and 24 km/h reduced PM_{10} emissions by 58 ± 3% and 42 ± 35%, respectively.

Gillies et al. (1999) and Watson et al. (1996b) describe a year-long study to assess changes in suppressant effectiveness in reducing PM_{10} emissions from unpaved public roads in California's San Joaquin Valley. Three different types of chemical suppressants were applied on the unpaved roads: an acrylic co-polymer, a bitumen with co-polymer additive, and a "bio-catalyst". Vehicle-induced vertical PM_{10} concentration profiles were measured.

Gillies et al. (1999) found that within a week of application the bitumen and acrylic co-polymer showed over 95% emission reduction as compared to the untreated surface. The "bio-catalyst" had an effectiveness of only 39%. After three months, the efficiency of the bitumen and bio-catalyst had decreased by 20% and 30%, respectively, while the efficiency of the acrylic co-polymer remained constant. After eleven months, the control effectiveness had been reduced to 53% for the bitumen, 85% for the acrylic co-polymer and 0% for the "bio-catalyst."

3.3.3 Street Cleaning

Mechanical broom and vacuum street sweeping are applied in urban areas to remove street debris, litter, and dirt rather than to control air pollution. Water droplets are often sprayed onto the road surface prior to sweeping to minimize dust resuspension induced by the sweeper. Sweeping schedules vary from weekly to monthly. Sweeping for aesthetic purposes is confined to the curb lanes in commercial and/or residential areas. Major highways and freeways are rarely swept.

Mechanical broom sweepers use large rotating brooms to lift the material from the street onto a conveyer belt. The conveyer discharges street debris into a collection hopper. Circular gutter brooms direct the debris into the path of the rotating broom. Mechanical broom sweeping has been discounted as a means of air pollution control on paved urban streets (Axetell and Zell, 1977; Gatz et al., 1983). Chow et al. (1990) found that the brushes resuspend as many small particles as they remove.

Vacuum sweepers use pure vacuum suction, regenerative-air suction, or blow-air/suction recirculation (Calvert et al., 1984; Duncan et al., 1985; Cowherd and Kinsey, 1986; Cowherd et al., 1988). A gutter broom loosens dirt and debris from the road surface and directs this material to a vacuum nozzle that sucks it into a hopper. The hopper usually consists of a chamber into which particles are collected by gravitational settling. The air passing through this chamber can be exhausted directly to the environment, through a bag-filter or precipitator, or to the collection nozzle for recirculation.

Pure vacuum sweepers create a strong vacuum within the pickup head that draws air from outside the head, through a duct, and into a hopper. The air movement across the road surface removes particles from the pavement and entrains them in the air flow. The vacuumed air is exhausted to ambient air after a brief residence time in the hopper. This residence time is usually insufficient to allow gravitational settling of particles in the PM_{10} size fraction.

Regenerative-air vacuum sweepers direct the exhaust air back to one end of the pickup head at velocities between 50 and 200 m/s. This blast air is directed perpendicular to the pavement where it is intended to dislodge dirt particles. The blast air and its entrained particles move across the pickup head to a suction nozzle that transports the debris to the collection hopper. The pickup head must seal with the pavement using a flexible rubber curtain to prevent blast air from escaping the collection nozzle and to maintain a negative air pressure within the nozzle.

The blow-air/suction recirculation sweeper directs a portion of the exhaust air to a blast nozzle located immediately behind the pickup head. This blast air is directed at an angle to the pavement to blow particles from the road surface into the airstream caused by suction through the head. The suction flow rate exceeds the blast flow rate so that ambient air is always being drawn into the pickup head, thereby minimizing the escape of recirculated particles to the air. The nonrecirculated portion of the exhaust air is vented into a separated settling chamber before it escapes to ambient air.

Water flushing uses pressurized sprays from a water truck to dislodge road dust and transport it to the curb, where much of the particulate is washed into the drain system. A combination of water flushing followed immediately by broom sweeping has been widely used as a means of street cleaning.

Water flushing generally results in more consistent and higher dust reduction than sweeping, with 30% to 80% control effectiveness determined by source measurements, and 0 to 18 $\mu g/m^3$ reductions in TSP determined by receptor measurements (Axetell and Zell, 1977; Cowherd, 1982; Cuscino et al., 1983b).

Flushing can be expected to reduce particle resuspension dramatically while the road surface is still wet, but its effect on average emissions during a typical cleaning cycle cannot be ascertained from the data reported. Results of the receptor-oriented studies are mixed and are based on TSP measurements. Flushing is generally considered to be more effective for reducing particle loadings on the street surface than broom or vacuum sweeping. There are few data to support this, and the results of studies based on TSP measurements are not directly applicable to PM_{10}.

The combination of water flushing and broom sweeping has the highest reported control effectiveness, from 47% to 90% for PM_{15} particles and 48% to 83% for $PM_{2.5}$ particles (Cowherd, 1982; Cuscino et al., 1983a, 1983b). However, the time frame of these measurements is either not specified or very short – less than three hours, when the road surface may still be wet. Effectiveness tests were made on industrial paved roads; applicability to public roads has not been demonstrated.

Seton, Johnson, and Odell, Inc. (1983) describe a six-month street-sweeping study conducted in Portland, OR, in 1981. Geological source contributions to chemically speciated TSP, PM_{15}, and $PM_{2.5}$ concentrations at a nearby sampling site were compared for sweeping and nonsweeping periods. No reductions in geological source contributions were detected with daily vacuum sweeping of the curb lane on industrial-area streets.

Hewitt (1981) reports a four-year study in Bangor, ME, where TSP averages during two years without regular sweeping periods were compared with averages over the following two years with a sweeping program in place. Nearly 800 TSP measurements were obtained and a 20% reduction in average TSP during the final study years was attributed to vacuum street cleaning.

PEDCo Environmental (1981) measured TSP, PM_{15}, and $PM_{2.5}$ concentrations near paved roads that had been sanded during wintertime snow storms in Denver, CO. One site

had the sand removed by sweeping while the other site had no sand removal. Though slight decreases in TSP and $PM_{2.5}$ were observed near the site where sand was removed, the difference was not statistically significant.

Cuscino et al. (1983b) made size-resolved vertical profile measurements near paved roads before and after sweeping. For PM_{15} mass concentrations, emission reductions of 16% were estimated at two hours after sweeping, 51% at three hours after sweeping, 0% at four hours after sweeping, and 58% at 24 hours after sweeping. Cuscino et al. (1983b) note that meteorological variability may have been responsible for these differences. In earlier studies, Cowherd (1982) reported efficiencies of about 45% for PM_{15} and 35% for $PM_{2.5}$ when vacuum sweeping was used. Both of these vacuum-sweeping studies were performed on industrial paved roads where initial street loadings and material tracked on is typically much higher than on public streets.

Chow et al. (1990) applied a receptor model to determine the contributions from PM_{10} dust and from primary motor vehicle exhaust near a roadway. The ratio of these two contributions is less sensitive to meteorological and emission rate differences between sweeping and nonsweeping study periods than the absolute source contributions from either source. Chow et al. (1990) found that daily street sweeping with a regenerative-air vacuum sweeper resulted in no detectable reductions in geological contributions to ambient PM_{10} measured in the sweeping area and that the street-sweeper design used in this study cannot possibly reduce PM_{10} on pavements.

Fitz and Bumiller (2000) evaluated the PM_{10} produced by different sweepers as well as their effectiveness in removing sandy material from a paved surface in a circus tent. Although more than 97% of the sand was removed from the roadway surface, the sweepers themselves generated from 7.5 to 160 mg/km of PM_{10} during the sweeping process. A modification of this test procedures was adopted by the South Coast Air Quality Management District (1999a, 1999b) as a fugitive dust emissions reduction rule.

While recently improved sweepers seem to effectively deplete the reservoir of material from which PM_{10} particles can be generated, none of these studies conclusively demonstrates the effectiveness of street sweeping on ambient concentrations of suspended PM_{10} after sweeping is completed. Chow et al. (1990) and Fitz and Bumiller (2000) indicate that the sweepers themselves generate substantial fugitive dust PM_{10} emissions. Suspended particle concentrations are affected by many variables and differences caused by street sweeping may not be detectable at nearby monitoring sites. The variables that appear to influence emission reduction efficiencies are: 1) loading of dirt on the street before and after sweeping; 2) particle size distribution of dirt on the street; 3) sweeper efficiency in removing dust from the street surface; 4) sweeper exhaust emission rates for small particles; 5) portion of roadway which is swept; 6) length of roadway that is swept; 7) sweeping frequency; and 8) meteorological variables such as precipitation, wind speed, wind direction, and relative humidity. The largest unknown variable for paved roads are the times and locations where the reservoir of suspendable dust is replenished. The previous studies offer little guidance on the quantitative effects of changes in these variables on geological contributions to PM_{10} or $PM_{2.5}$ concentrations measured near paved roads.

3.3.4 Surface Roughness Alterations

Fugitive dust emissions introduced by wind erosion can be mitigated by: 1) altering the surface (such as increasing its roughness with tillage implements and practices), 2) providing sufficient vegetative stubble or mulch coverage, or 3) using windbreaks. Applying tillage implements to ridge the soil or create large nonerodible clods is a standard practice in wind erosion control where the establishment of vegetative coverage is difficult. This method results in large nonerodible roughness elements that absorb momentum from the erodible soil and trap the erodible soil in larger cracks and crevices.

No-till and minimum-till planting procedures have been developed to minimize topsoil erosion, conserve water, and reduce dust emissions (Rice, 1983). During farming practice, new seeds are planted into stubble from the last crop or into a cropped field by cutting through the surface vegetation and inserting the seed directly into the ground. This type of low- or no-till farming operation reduces wind-generated fugitive emissions because vegetative cover (e.g., standing stubble or mulch) is left on the soil surface. Vegetation protects the soil by absorbing the energy of the surface wind, reducing the wind velocity, and sheltering the resuspendable particles. The fraction of vegetative cover, its shape, and density of coverage determine the effectiveness of wind-blown dust controls (Leys, 1991). Annual crops can be interplanted in narrow strips or rows to minimize ground-level soil movement. Interplanting is often supplemented with other practices such as protecting the strips with other vegetative coverage to more efficiently reduce dust emissions.

Windbreaks or other surface barriers absorb or deflect wind energy and minimize soil movement or particle resuspension from unprotected surfaces. The length of a windbreak needs to be six to ten times its height to effectively mitigate dust emissions. In general, the shape, width, height, and porosity of the barrier, along with the surface wind speed and wind direction, affect the efficiency of the windbreak.

3.3.5 Minimization of Activity

Fugitive emissions can be reduced by minimizing the activities that generate them. Vehicle-related dust can be reduced by regulating vehicle travel speeds, prohibiting road use by vehicles with more than four wheels, and reducing vehicle volume by providing perimeter parking and mass transit to industrial sites. Traffic controls are often supplemented with other control measures, such as surface watering and wheel washing, to minimize dust emissions induced by anthropogenic activities.

3.4 Emission Estimates

The effects of different selections for each of the variables in Equation 3-1 and the different levels of spatial and temporal distributions become apparent when fugitive dust emissions from the National Trends Inventory of Table 2-1 are compared with a state-generated inventory that has more specific information available to it.

Table 3-2 compares recent countywide inventories compiled by the California Air Resources Board (ARB) with those determined from the National Trends $PM_{2.5}$ inventory (U.S. EPA, 1998a) for unpaved road, paved road, and construction emissions. The national inventory makes assumptions for emission factors and uses activity databases that are common to all states and localities. This assures that there is a consistency among all emission estimates and that biases between emissions from different states are minimal. National emission estimates are usually made for an entire state and are apportioned to counties in proportion to a reasonable surrogate (e.g., population, area, miles of road).

State and local inventories such as those from California may use different sources of activity data, different emission factors, and in some cases wholly different emission estimation methodologies than were used in the preparation of the national emissions inventory.

Most local emissions inventories begin with the default values in EPA's AP-42 (U.S. EPA, 1999). These are improved by substituting local activity data and, when available, locally derived emission factors which reflect specific local conditions. For unpaved road, paved road, and construction emissions, Table 3-2 shows that ARB's fugitive dust emission estimates are 59% of EPA's estimates. This large difference is within the expected uncertainty of most of the emission estimates. Ratios of ARB to EPA emission rates by county range from 7% for San Francisco County to 2,800% for Inyo County. Explanations for these discrepancies offer insight into reasons why the proportions of fugitive dust emissions in inventories do not compare well with geological source contributions to ambient PM_{10} and $PM_{2.5}$ samples.

The national inventory uses a construction emission factor that is about ten times higher than the emission factor used in the California inventory. The California emission factor was developed from recent field analysis of construction sites in the western United States (Midwest Research Institute, 1996). For paved roads, the national inventory assumes 79% control in nonattainment areas due to street sweeping and incorporates a moisture factor that also reduces emissions. ARB does not reduce emissions due to street sweeping and does not include a moisture reduction. However, the EPA inventory emission factors are two to five times higher than the California emission factors that use silt loadings specific to California roads. Vehicle miles traveled on paved and unpaved roads are also determined from different activity data sets. Details about these differences are discussed below.

Knowledge about the counties included in Table 3-2 and the urban areas in those counties that are monitored for PM_{10} and $PM_{2.5}$ provides additional insight into the discrepancies. Fresno and San Francisco counties both contain large urban areas of about the same spatial extent (the cities of Fresno and San Francisco). While the city of San Francisco completely occupies San Francisco County, the city of Fresno occupies only a few percent of the land area in Fresno County that extends from the coastal mountains, through the San Joaquin Valley, and to the crest of the Sierra Nevadas. According to Table 3-2, the ARB and EPA inventories correctly assign no unpaved road emissions to highly urbanized San Francisco County. Fresno County contains numerous miles of agricultural and back-country unpaved roads for which the traffic volumes are uncertain. The effect on emissions of

Table 3-2. Comparison of California Air Resources Board (ARB) and U.S. EPA $PM_{2.5}$ 1995 emissions by county.

	$PM_{2.5}$ Emissions (tons per year)								ARB/
	Unpaved Road		Paved Road		Construction		Total		EPA
County	ARB	EPA	ARB	EPA	ARB	EPA	ARB	EPA	(%)
ALAMEDA	51	38	720	2152	261	5894	1032	8084	13
ALPINE	102	8	5	7	3	0	110	15	733
AMADOR	235	155	30	126	14	22	279	303	92
BUTTE	701	213	169	308	81	383	951	904	105
CALAVERAS	393	218	31	177	23	79	447	474	94
COLUSA	574	88	49	72	3	1	626	161	389
CONTRA COSTA	94	111	512	1393	301	3921	907	5425	17
DEL NORTE	268	118	22	96	25	11	315	225	140
EL DORADO	527	546	198	449	80	383	805	1378	58
FRESNO	2801	357	838	338	303	1484	3942	2179	181
GLENN	428	106	38	88	4	8	470	202	233
HUMBOLDT	794	344	122	288	35	205	951	837	114
IMPERIAL	1712	173	198	164	114	80	2024	417	485
INYO	1418	29	46	17	3	6	1467	52	2821
KERN	2119	278	604	236	302	910	3025	1424	212
KINGS	599	98	111	57	49	30	759	185	410
LAKE	438	244	49	200	30	49	517	493	105
LASSEN	1290	157	58	128	6	8	1354	293	462
LOS ANGELES	946	243	5838	2716	1135	15865	7919	18824	42
MADERA	575	124	160	72	40	91	775	287	270
MARIN	81	111	151	90	30	17	262	218	120
MARIPOSA	425	121	25	455	5	921	455	1497	30
MENDOCINO	878	423	109	346	65	134	1052	903	117
MERCED	1039	340	289	387	89	203	1417	930	152
MODOC	1068	50	23	41	2	2	1093	93	1175
MONO	1507	37	33	34	4	34	1544	105	1470
MONTEREY	599	468	308	780	97	611	1004	1859	54
NAPA	58	146	70	245	29	385	157	776	20
NEVADA	519	413	98	338	109	212	726	963	75
ORANGE	38	12	1925	715	337	8099	2300	8826	26
PLACER	382	454	265	520	199	858	846	1832	46
PLUMAS	857	119	33	97	20	4	910	220	414
RIVERSIDE	1009	504	1982	591	362	2879	3353	3974	84
SACRAMENTO	633	229	741	539	499	2805	1873	3573	52
SAN BENITO	428	135	39	112	26	71	493	318	155
SAN BERNARDINO	2363	318	1881	615	842	2775	5086	3708	137
SAN DIEGO	1246	930	1764	4881	1850	9577	4860	15388	32
SAN FRANCISCO	0	0	236	1205	52	3174	288	4379	7
SAN JOAQUIN	903	213	504	216	173	960	1580	1389	114
SAN LUIS OBISPO	707	329	236	359	247	594	1190	1282	93
SAN MATEO	116	55	359	1114	172	3040	647	4209	15

Table 3-2. (continued)

	$PM_{2.5}$ Emissions (tons per year)								ARB/
	Unpaved Road		Paved Road		Construction		Total		EPA
County	ARB	EPA	ARB	EPA	ARB	EPA	ARB	EPA	(%)
SANTA BARBARA	244	183	248	641	217	1173	709	1997	36
SANTA CLARA	692	242	821	2810	311	6609	1824	9661	19
SANTA CRUZ	342	208	148	494	129	539	619	1241	50
SHASTA	1040	441	146	622	66	529	1252	1592	79
SIERRA	683	26	12	21	2	0	697	47	1483
SISKIYOU	968	232	99	190	15	22	1082	444	244
SOLANO	344	144	244	688	131	1362	719	2194	33
SONOMA	1098	830	224	1067	107	1778	1429	3675	39
STANISLAUS	440	165	382	158	306	825	1128	1148	98
SUTTER	394	119	66	183	24	126	484	428	113
TEHEMA	588	243	72	199	13	40	673	482	140
TRINITY	86	55	18	45	68	0	172	100	172
TULARE	1157	309	482	204	134	372	1773	885	200
TUOLUMNE	298	274	55	224	33	129	386	627	62
VENTURA	222	241	495	1289	242	1865	959	3395	28
YOLO	483	118	129	258	447	500	1059	876	121
YUBA	260	125	52	173	11	34	323	332	97
Total	**40,260**	**13,010**	**24,562**	**32,030**	**10,277**	**82,688**	**75,099**	**127,728**	**59**

mountain and desert roads, many of which are maintained by state and national forest or land management services, can be seen in emissions from Lassen, Inyo, Mono, Placer, and other mountain/desert counties. Mountain and desert roads are distant from population centers where PM is monitored.

Paved road dust is the dominant fugitive dust emission that would affect San Francisco and Fresno particle monitors. ARB's estimate is only one-fifth of the EPA estimate for San Francisco County, but more than twice EPA's estimate for Fresno County, Given that Fresno County includes several highly traveled freeways linking the southern and northern parts of the state (I-5 and SR-99), while San Francisco traffic is congested and primarily local, the ARB proportions are more reasonable than the EPA values. Paved road dust emissions within the Fresno city limits would probably be less than 20% of the countywide total. EPA construction estimates for San Francisco are nearly twice those for Fresno, even though San Francisco is completely built out and the cities in Fresno County are among the most rapidly growing urban areas in the state. The ARB estimates are more consistent with this in magnitude and proportion between the two cities.

More detailed comparisons of the methods used by EPA and ARB, and the differences between them, are given in the following sub-sections for different fugitive dust source categories.

3.4.1 Unpaved Road Dust Emissions

Emission factors used for the National Trends Inventory (U.S. EPA, 1998b) presented in Table 2-1 is given below. These are consistent with the road dust emissions included in the PART5 mobile source emission model (U.S. EPA, 1995):

$$R_{ur,TSP} = 5.9 \times (Silt/12) \times (Speed/30) \times (Weight/3)^{0.7} \times (Wheels/4)^{0.5} \qquad (3\text{-}2)$$

where:

$R_{ur,TSP}$ = TSP unpaved road dust emission factor for all vehicle classes combined (grams per mile).

Silt = silt fraction content of the surface material (% mass).

Speed = average speed of all vehicle types combined (miles per hour [mi/hr]).

Weight = average weight of all vehicle types combined (tons).

Wheels = average number of wheels per vehicle for all vehicle types combined.

The most recent AP-42 emission factor (U.S. EPA, 1999) eliminates the speed factor and is specific to PM_{10}, but this factor is being re-evaluated and has not yet been used to estimate annual trends.

For 1995 national estimates, average unpaved road silt fractions were compiled for the 1985 National Acid Precipitation Assessment Program by the Illinois State Water Survey. The Survey measured silt from ~200 unpaved roads in thirty states. Average silt contents of unpaved roads were calculated for each state that had three or more samples for that state. For states that did not have three or more samples, the average for all samples from all states was substituted.

Vehicle speeds were assumed to be 20 mi/hr for urban local, arterial, and collector roads. For non-urban roads, speeds were 39 mi/hr for minor arterials, 34 mi/hr for major collectors, and 30 mi/hr for minor collectors and local roads. Estimates of average vehicle weight and average number of wheels per vehicle over the entire vehicle fleet were obtained from the Motor Vehicle Manufacturers Association (1991), Crain Communications Inc. (1991), and the U.S. Department of Transportation.

The control measure modifier consists of two parts. The first part assumes that no emissions occur during rain: $P_{annual,state}$=(365-PrecipDays)/365 where PrecipDays is the number of precipitation days per year with greater than 0.01 inches of rain. Days of precipitation for each state were obtained from the National Climatic Data Center, averaging over several meteorological stations within each state. The second part assumes that one of the watering or chemical stabilization procedures described above is applied in areas that exceed PM_{10} standards. On urban unpaved roads in moderate PM nonattainment areas, paving unpaved roads is assumed. This control was applied with a 96% effectiveness to 50% of unpaved roads. On rural roads in serious PM nonattainment areas, chemical stabilization was assumed. This control was applied with a 75% effectiveness to 50% of unpaved roads. On urban unpaved roads in serious PM nonattainment areas, paving and chemical

stabilization controls were assumed with an aggregate effectiveness of 90% on 75% of the road surfaces.

Particle size modifiers for PM_{10} and $PM_{2.5}$ are $K_{ur,PM10}$= 0.36 and $K_{ur,PM2.5}$=0.0525.

The activity factor ($A_{annual,state}$) database was for vehicle miles traveled (VMT) from average daily traffic volumes and miles of unpaved roads for county and non-county (state or federally) maintained roadways. The U.S. Department of Transportation reports state-level, county-maintained roadway mileage estimates by surface type, traffic volume, and population category. From these data, state-level unpaved roadway mileage estimates were derived for the volume and population categories.

State and federally maintained unpaved road VMT were added to the county-maintained VMT for each state and road type to determine each state's total unpaved road VMT by road type. The state-level unpaved road VMT by road type were then allocated by month with temporal allocation factors from the National Acid Precipitation Assessment Program. Monthly state-level, road type-specific VMT were multiplied by the corresponding monthly, state-level, road type-specific emission factors. The state/road type-level unpaved road PM emission estimates were allocated to each county in the state using estimates of county rural and urban land area from the U.S. Census Bureau for the years 1985 through 1989. For the years 1990 through 1996, 1990 county-level rural and urban population was used to distribute the state-level emissions instead of land area.

The ARB procedures used substantially different emission factors, activities, and emission reductions. California-specific PM_{10} emission factors (Flocchini et al., 1994; Watson et al., 1996b; Gillies et al., 1999) were used, resulting in 2.27 lbs PM_{10}/VMT, lower than the previous emission factors derived using the AP-42 methodology. California Department of Transportation data were used to estimate the miles of unpaved road in each county. Unpaved road mileage in agricultural areas was estimated as a fraction of crop acreage. Agricultural road travel was estimated as 175 VMT/40-acre parcel.

No emission controls were assumed as there is little information on how much this is used in California, and the long-term effectiveness of most suppressants has not been proven (Watson et al., 1996b). Effects of precipitation were estimated monthly using regional rainfall data. During wet months when unpaved roads are muddy, it was assumed that VMT was lower along with lower emissions per VMT.

Even with greater specificity for California, the unpaved road emission estimates are uncertain. Using a single emission factor for all unpaved roads does not adequately represent the diversity of unpaved roads in California. In addition, estimates of VMT on unpaved roads are uncertain. Lacking better information, ARB assumes that each mile of unpaved road receives ten vehicle passes per day.

3.4.2 Paved Road Dust

U.S. EPA (1998) emission factor used for the National Trends Inventory presented in Table 2-1 is given below. This is consistent with the road dust emissions included in the PART5 mobile source emission model (U.S. EPA, 1995):

$$R_{pr,size} = K_{size} \times (SiltLoad/2)^{0.065} \times (Weight/3)^{1.5} \quad (3\text{-}3)$$

where:

$R_{pr,size}$= PM_{10} or $PM_{2.5}$ road dust emission factor for all vehicle classes combined (grams per mile).

K_{size}= 7.3 g/mi for PM_{10} or 1.8 g/mi for $PM_{2.5}$.

SiltLoad=silt loading of the surface material (g/m^2).

Weight =average weight of all vehicle types combined (tons).

Note that the size fraction is incorporated into the emission factor and that the silt loading rather than the fraction of silt is used. There is no dependence of the paved road dust emission factor on vehicle speed or number of wheels as there is in Equation 3-2 for unpaved road emissions.

For the national inventory, one of three paved road silt loadings is assigned to twelve functional roadway classifications (six urban and six rural) based on the average annual traffic volume of each functional system by state. Values of 1 g/m^2 were assigned to local functional class roads. A silt loading of 0.20 g/m^2 was assigned to road types that had average traffic volumes less than 5,000 vehicles/day, and a loading of 0.04 g/m^2 was assigned to road types that reported more than 5,000 vehicles/day. Average daily traffic volumes were determined by dividing annual VMT for each state and functional class by state-specific functional class roadway mileage.

The unpaved road dust precipitation effectiveness factor was applied to paved road dust using the same National Climatic Data center information for 1995 and 1996 estimates. The assumed control was vacuum sweeping of paved roads twice per month with an effectiveness of 79%. This control was assumed for urban and rural roads in serious PM nonattainment areas and for urban roads in moderate PM nonattainment areas. The diligence of sweeping varied by road type and attainment classification (serious or moderate).

State/road-type-level VMT for paved roads was calculated by subtracting the state/road-type-level unpaved road VMT from total state/road-type-level VMT. There are cases where unpaved VMT is higher than total VMT because estimation methods differ. For these cases, unpaved VMT was reduced to total VMT and paved road VMT was assigned a value of zero. The paved road VMT were allocated by month using the NAPAP temporal VMT allocation factors. These monthly/state/road type-level VMT were multiplied by the corresponding paved road emission factors. Paved road emissions were allocated to counties according to the fraction of total VMT in each county for the specific road type.

ARB paved road dust emissions are estimated for freeways, major streets/highways, collector streets, and local streets. California-specific silt loadings were used with county-specific VMT on each of the four roadway types. Emission growth for future years was changed so that freeways and major road travel grows in proportion to increases in roadway centerline mileage, and local and collector road travel grows in proportion to VMT increases. Previous travel projections were based solely on VMT.

Incorporation of all of these changes reduced California paved road dust emission estimates by about 70% from the previous published inventory values. California specific roadway silt loadings were the major cause of this reduction; U.S. EPA (1999) includes many silt loading samples from other states (predominantly Montana) that have road sanding and other soil sources.

3.4.3 Construction

The national inventory uses the following emission factor that was determined from upwind and downwind emissions around selected construction sites:

$$R_{const,TSP} = 1.2 \text{ tons/acre/month of activity} \tag{3-4}$$

A size factor of 0.22 is applied for PM_{10} emissions. Dust control effectiveness of 62.5% for PM_{10} and 37.5% for $PM_{2.5}$ were applied to counties classified as PM nonattainment areas.

The acres of land under construction were estimated by EPA region based on the U.S. Census Bureau's 1987 dollars spent on construction. These estimates are available every five years. Regional-level emissions were distributed to the county-level using county estimates of payroll for construction from County Business Patterns.

For the California construction dust inventory, an improved emission factor (Midwest Research Institute, 1996) reduced emission estimates by ~70%. The construction dust source category includes fugitive dust particulate matter emissions caused by construction activities while building residential structures, commercial structures, and roads. The estimated emissions result from individual construction operations such as scraping, grading, loading, digging, compacting, light-duty vehicle travel, and other operations. Only the heavy earth-moving portion of a construction project approaches the emissions indicated by Equation 3-4, and this is a relatively short fraction of the duration of the project. For most parts of California a value of 0.11 tons of PM_{10}/acre-month is used. Activity data are similar to those used by the national inventory.

3.4.4 Wind Erosion

Since wind erosion depends on wind speeds at the surface exceeding the threshold velocity, hourly wind speed measurements at a known height above ground level are needed. Gusts or "fastest mile" winds are used in the estimation method. Gusts are the highest speed recorded for 60 seconds or less during an hour. The fastest mile is reported by the National Weather Service as the speed corresponding to the fastest wind within an hour to traverse a full mile. The emission factor for wind erosion is:

$$R_{eros,PM10} = 0.5 \times \Sigma(58(u^* - u_t^*)^2 \times 25 \times (u^* - u_t^*) \tag{3-5}$$

$$\text{Sum only for } (u^* - u_t^*) > 0$$

where:

$R_{eros,PM10}$ = PM_{10} emission factor (g/m^2 of available reservoir).

u^* = friction velocity at surface (m/s).

u_t^* = threshold suspension velocity at surface (m/s).

For a typical configuration with 0.05 surface roughness and measurements from a 10 m meteorological tower, the surface friction velocity is ~0.053 times the gust or fastest mile speed measured at 10 m. U.S. EPA (1999) provides several examples and default values for surface roughness that can be used to adapt the emission factor to a variety of landforms. Particle size modifiers are $K_{eros,PM10}$=1.0 and $K_{pr,PM2.5}$=0.30.

The national inventory estimates wind erosion from annual average wind speeds applied to land use data (Barnard and Stewart, 1992). Most of these emissions derive from tilled agricultural land. Urban windblown dust is not included in the national inventory.

The California inventory also emphasizes agricultural windblown dust rather than urban windblown dust, and wind erosion of desert areas is not included in the inventory. Climatological data used by the soil erosion equation was improved by estimating region specific temperature, rainfall, and wind speed by month. This approach gives a much better representation of the seasonality of windblown dust emissions.

Emission estimates do not explicitly account for irrigation effects, which tend to reduce emissions. The ARB inventory includes the dust-reducing effects of irrigation by treating it as a form of precipitation. The adjustment takes into account the overall soil texture, number of irrigation events, and the fraction of wet days during the time period. The existing wind erosion equation also does not take into account the effect of growing crop cover on windblown dust emissions. Using information gathered from agricultural experts, monthly crop canopy profiles were developed for the major California crops. Wind erosion potential for the soil under each crop was modified based on the quantity of vegetation available to protect the soil from erosion as the growing season progresses. The updated estimates also include factors for the post-harvest soil cover, post-harvest planting, and the amount of bare and unplanted border area for major California crops.

California windblown dust estimates are estimated for each month using location specific soils, climatic, and land use data. The inventory also incorporates crop specific irrigation, crop coverage, farming practice, and land use data into the windblown dust equation. These methodological improvements reduced the PM_{10} windblown dust estimates by ~80% from previous estimates.

3.4.5 Active Storage Piles

Active storage piles generate dust when material is dropped onto them from above in an unenclosed area. Material can be added in batches or continuously. The material is carried away from the pile by wind, so this is an important variable in emission estimation. The emission factor for storage piles is:

$$R_{pile,PM10} = .0011 \times (\text{WindSpeed}/5)^{1.3} \times (\text{Moisture}/2)^{-1.4} \qquad (3\text{-}6)$$

where:

$R_{pile,PM10}$ = PM_{10} storage pile emission factor (pounds/ton of material dropped).

WindSpeed = mean wind speed (m/hr).

Moisture = material moisture content (%).

Particle size modifiers are $K_{pile,PM10}$=1.0 and $K_{pile,PM2.5}$=0.314. Multiply $R_{pile,PM10}$ by 0.5 to obtain units of kg/megagram of material dropped. Silt content was not found to be a significant variable in the regression analysis used to generate Equation 3-6. The range of conditions for the emission tests that generated Equation 3-6 were silt contents of 0.44% to 19%, moisture contents of 0.25% to 4.8%, and wind speeds of 1.3 to 15 mph. These emissions are a very small fraction of the EPA and ARB inventories.

3.4.6 Other Fugitive Dust Sources

Several other sources contribute to fugitive dust emissions, but they are not normally quantified in inventories. These include landfills, leafblowers, equestrian facilities, turbulent wakes from vehicles, sod overseeding, uncovered haul trucks, and urban wind erosion. Botsford et al. (1996) examined these emissions and found them minor compared to those from paved roads, unpaved roads, construction, and wind erosion. These may be important within certain neighborhoods, however.

In urban areas, solid wastes are disposed of in landfills. Daily accumulations of waste material are dumped in an active area of the landfill until daily maximum input is achieved. At the end of the disposal day, earthen materials, or sometimes green waste materials, are used as a covering to the daily waste material. Soils are typically hauled in to the site to be used as the daily cover. At some landfill sites, soils are excavated from adjacent areas; at other landfills, soils may need to be hauled in from more distant locations. Because the location of the daily solid waste dumping can vary from day to day, temporary unpaved roads are used by the solid waste haul trucks. Fugitive dust sources include unpaved road dust; reentrained road dust on paved road portions of landfills or adjacent roadways from trackout; soil handling, both from the excavation portion and the cover portion; and windblown dust.

Botsford et al. (1996) examined the potential for fugitive dust emissions from five landfills in California and four in Nevada. Overall emissions were determined to be a sum of three different fugitive dust components: road emissions (paved and unpaved), working emissions (excavation and cover), and wind erosion. Botsford et al. (1996) found landfill emissions to range from less than one ton per year for a small landfill about 0.6 km^2 and handling about 1,500 m^3 of waste material per day to nearly 38 tons per year for a larger landfill about 5 km^2 in size and handling about 8,000 m^3 of waste per day. It was concluded that landfills accounted for less than 1% of the local urban area fugitive dust emissions inventory in Las Vegas and Los Angeles.

Leafblowers are used primarily by homeowners and gardeners. Often cited as a local nuisance due to noise and deposited particles, some municipalities in Southern California have moved to ban their use. Some air quality regulations foster cleaner burning leafblowers to reduce VOC emissions from gas powered machines, but do little to reduce fugitive dust.

Botsford et al. (1996) made an aerodynamic calculation of potential dust suspension and concluded that about 420,000 units were in operation in the South Coast Air Basin (SoCAB). Assuming that, on average, each leafblower is used one day per week, and that the area cleaned is approximately 1,000 m^2, fugitive dust emissions could be 9 tons per day. This represents ~2% of the 1993 SoCAB emissions inventory.

Even in urban areas, recreational horseback riding is popular, and equestrian centers offer riding opportunities for enthusiasts, including riding trails. At faster paces, horses traveling over dry dusty surfaces can kick up clouds of dust. Botsford et al. (1996) constructed a simple box model to estimate emissions from horse-riding activities. Based on surveys of equestrian centers in the Los Angeles area, it was concluded that about 13,000 horses were associated with equestrian centers, and that PM_{10} emissions from horse-riding activities were less than 0.2 tons per day, or less than 0.1% of the SoCAB fugitive dust inventory.

In some locations, travel lanes abut unpaved shoulders or unpaved sidewalk areas (even where curbing may exist). It has been observed that large vehicles, such as trucks and buses can cause dust plumes from the adjacent unpaved surfaces due to turbulent wake effects. Botsford et al. (1996) attempted to estimate the magnitude of this source by adjusting the paved road emission factor to account for silt content on the unpaved shoulder. The wake velocity, or turbulence intensity, was represented as a function of the distance from the side of the truck. Unpaved shoulder emissions in California's Coachella Valley were estimated to be ~2.3 tons per day. Countess and Countess (1996) noted an error in the VMT data and concluded that using the approach taken, the emissions from this source would be the single largest source category for the area. Moosmüller et al. (1998) found only the wakes behind large trucks near unshouldered roads were sufficient to suspend dust from the shoulder. Their estimates were a relatively small amount of total dust emission estimates for California's San Joaquin Valley.

Western golf courses and other recreational areas that need to maintain green grass throughout the year overseed annual grasses that go dormant during the winter. This process ceases watering over a two- to three-week period prior to overseeding in order to enhance the dormancy process. Once dormant, special raking machines scour the sod to collect the residual dormant grass (thatch), and to apply fresh seed. This process is extremely dusty, and in California's Coachella Valley it results in noticeable increases in measured PM_{10} levels during the fall. No emission estimates have been made for this source, although emission reduction measures have been tested by Fitz (1995). Aside from the Coachella Valley, other similar areas have not been identified. This is likely to be important in only a very few urban areas, and the need for national inventories for this source is not deemed necessary.

Despite the high degree of finished surfaces (e.g., pavement, buildings, landscaping) in urban areas, there are ample opportunities for mechanical and windblown fugitive dust emissions. Available surfaces include vacant properties, unpaved alleyways, unpaved areas in commercial/industrial facilities, and unmaintained roadway median strips. Mechanical disturbances are caused by vehicles traveling over these surfaces, and because such surfaces are not considered to be unpaved roads, emissions are not calculated. In urban areas, buildings cause additional turbulence that can enhance dust suspension even under relatively

low wind speeds. There are no known inventories to determine the extent to which such surfaces exist within urban areas, nor have there been any estimates of the importance of fugitive dust emissions from these sources. Some preliminary surveys and emission estimates are needed to ascertain the importance to overall urban fugitive dust emission inventories.

Trucks carrying aggregate material can be a source of emissions when traveling at speeds that can cause emissions to occur. Mitigation actions are required in many areas, including covering or adequate freeboard (e.g., the top of the hauled load must be lower than the sides of the container). In most circumstances, the main effects from haul trucks are the losses of larger materials that contribute to paved road silt loadings, and are inherently accounted for within the paved road dust emission categories. Nevertheless, there may be direct PM emissions as well.

3.5 Hypothesis Testing

This section provides information on the third, fourth, and fifth hypotheses advanced in Section 1.

- **Incompatible temporal and spatial averaging of fugitive dust emissions relative to ambient PM measurements.** The comparison between California and national emission estimates shows substantial differences by county. Some of the differences are due to the national inventory's allocation of statewide activities to counties as compared to California's aggregation of countywide activities to the state level. Differences in assumed silt loadings are a major cause of differences for road dust. Unpaved road mileage and VMT are other causes. Differences are also due to averages across large counties that contain small but highly populated urban areas imbedded in large but sparsely populated non-urban areas. Most of the PM ambient measurements are taken within the population centers and do not represent county-average concentrations.
- **Inaccurate formulation of emission factors.** Construction factors in the national inventory clearly overestimate actual construction emissions. They should apply only to those portions of the construction project that involve earth moving. The California construction emissions are considerably lower because they are adjusted for emissions during the different phases of a construction project. Area-specific silt loadings and silt fractions also reduce emission factors in California relative to the values used in the national emissions inventory.
- **Insufficient and uncertain activity levels, specifically with respect to the reservoir of suspendable particles, particle size distributions in the reservoir, meteorological variables, and human intervention.** The county specific activity data available in California better allocate emissions than the statewide activity databases used in the national inventory. On the other hand, the national inventory assumes control effectiveness values that are not borne out by fugitive dust control demonstration studies. There are still large uncertainties and variabilities in the activity databases for both inventories.

4. MODELS FOR FUGITIVE DUST CONTRIBUTIONS

Estimates of source contributions from fugitive dust and other sources are made using air quality models. Source-oriented emissions, meteorology, chemical transport models, and receptor models are often used independently to assess source contributions. A better integration of these different modeling systems would identify emission mechanisms that are not adequately represented in any of the models and would improve the attribution of excessive PM_{10} and $PM_{2.5}$ concentrations to fugitive dust sources.

Emission models need to be compatible with the spatial scales of atmospheric dispersion models and PM_{10} or $PM_{2.5}$ measurement networks. These scales (Watson et al., 1997b) are given below. Distances indicate the diameter of a circle, or the length and width of a grid square, with a monitor at its center.

- **Microscale (10 to 100 m):** Microscale pollutant concentrations show significant differences between locations separated by 10 to 50 m. This often occurs next to a emission sources, such as a busy roadway, construction site, vent, or short stack.

- **Middle Scale (100 to 500 m):** Middle-scale pollutant concentrations show significant differences between locations that are ~0.1 to 0.5 km apart. Middle-scale zones of representation are often source-dominated.

- **Neighborhood Scale (500 m to 4 km):** Neighborhood-scale pollutant concentrations do not show significant differences with spacing of a few kilometers. This dimension is often the size of emission and modeling grids used in large urban areas for source assessment. Sources affecting neighborhood-scale sites typically consist of small individual emitters, such as clean, paved, curbed roads, uncongested traffic flow with a small number of heavy-duty vehicles, or neighborhood use of residential heating devices such as fireplaces and wood stoves.

- **Urban Scale (4 to 100 km):** Urban-scale monitors show consistency among concentrations separated by of tens of kilometers. These concentrations represent a mixture of pollutants from many sources within an urban complex, including those from the smaller scales.

- **Regional-Scale Background (100 to 1,000 km):** Regional-scale concentrations show consistency among measurements for separations of a few hundred kilometers. Regional concentrations are often more consistent for secondary pollutants, such as ozone, sulfate, and nitrate, than they are for primary PM_{10} or $PM_{2.5}$ emissions. Regional-scale concentrations are a combination of naturally occurring substances as well as pollutants generated in urban and industrial areas that may be more than 1,000 km distant. Regional-scale sites are best located in rural areas away from local sources, and at higher elevations.

- **Continental-Scale Background (1,000 to 10,000 km):** Continental-scale background concentrations show little variation even when they are separated by

more than 1,000 km. They are hundreds of kilometers from the nearest significant emitters. Although these sites measure a mixture of natural and diluted manmade source contributions, the manmade component is at its minimum expected concentration.

- **Global-Scale Background (>10,000 km):** Global-scale background concentrations represent source emissions transported between different continents as well as naturally emitted particles and precursors from oceans, volcanoes, and windblown dust.

4.1 Emission Models

Previous sections have discussed the emissions inventory as a static entity based on empirical factors related to commonly available data. While emissions inventories provide useful compilations of information, they will not achieve their full potential until they are elevated to the status of other air quality models that are applicable to specific situations. This is especially true for fugitive dust emissions, as these are not confined specific times and locations as are industrial sources and vehicle exhaust.

Several types of emission models are considered separately: 1) speciated rollback; 2) horizontal and vertical flux; and 3) dynamic activities.

4.1.1 Speciated Rollback

Linear rollback (Barth, 1970; deNevers and Morris, 1975; Cass, 1981; Cass and McRae, 1981, 1983) is the most commonly used method for control strategy development, although it is not often identified as such. Rollback assumes that atmospheric concentrations in excess of background are proportional to aggregate emission rates. Reducing excessive concentrations of a pollutant to levels below a pre-set standard requires emission reductions that are proportionally equal to the relative amount by which the standard is exceeded, after background subtraction. Linear rollback for PM_{10} or $PM_{2.5}$ mass emissions identified in Table 2-1 nearly always targets fugitive dust sources because these are the largest mass emitters.

A better approach is a speciated rollback that applies to the major chemical components of PM_{10} or $PM_{2.5}$ (i.e., geological material, carbon, sulfate, nitrate, and ammonia). This requires a speciated inventory that applies chemical profiles to emission estimates from each source type. Speciated inventories have only been created for specific modeling studies and are not commonly available in non-attainment areas.

Speciated linear rollback has the fewest complex data requirements, but it also carries large uncertainties. These uncertainties might not, however, be larger than those associated with other modeling methods. They may be acceptable for selecting among different pollution control measures, or at least narrowing the scope of viable alternatives.

Linear rollback does not consider the effects of meteorological transport between source and receptor or of differences in gas-to-particle conversion for different precursor

emitters. It is most valid for spatial and temporal averages of ambient concentrations that represent the entire airshed containing urban-scale sources. The effect of transport from distant sources located outside the airshed is compensated by subtracting background concentrations, measured nearby but outside the airshed, from ambient levels prior to determining needed emission reductions. Linear rollback also assumes for secondary particles, such as ammonium nitrate and ammonium sulfate, that one of the precursors limits particle formation.

Whether or not speciated linear rollback is used to develop control strategies, a speciated emission model is needed to better evaluate emission estimates against ambient concentrations. Over year-long temporal and spatial averaging times, the ratio of carbon to geological concentrations above background levels should be similar in air samples and in a speciated primary particle inventory.

4.1.2 Horizontal and Vertical Fluxes

Fugitive dust emissions are emitted over large surface areas and are not ducted. This makes their quantification inherently less accurate than emission tests on stacks or other ducts through which effluents are ejected into the air.

Fugitive dust emission factors are most commonly determined by measuring the horizontal flux from an emitting area such as a road or vacant lot (Cowherd et al., 1984). This is accomplished by locating sampling systems with the desired size-selective inlet (TSP, PM_{10}, or $PM_{2.5}$) at various elevations downwind of the dust-emitting area. Monitors located upwind are used to determine the flux into the emitting domain.

Each of the downwind samplers is used to represent the amount of dust carried by the wind component perpendicular to a plane parallel to the source. Both the wind speed and concentration vary with height above ground level, so the horizontal flux is calculated through an area that extends above and below each sampler. These fluxes are added to obtain the aggregate emission rate from the source, after the flux of particles into the emitting area has been subtracted. Figure 4-1 shows a typical configuration for making these measurements. Sampler elevations, distances from the source, and measurement devices vary among the different studies that have been conducted.

Figures 4-2 through 4-4 show the cumulate horizontal emission fluxes at different elevations above ground level (agl) for paved roads, unpaved roads, and bare soil/construction sites. These plots result from many the same downwind profile tests (Cowherd, 1999) that were used to derive commonly used PM_{10} emission factors. The most noticeable feature from these plots is that approximately 60% to 90% of the horizontal emission flux is detected at elevations less than 2 m agl.

A 10 µm aerodynamic diameter particle has a settling velocity of ~0.3 cm/s, and would therefore deposit to the surface within ~5 minutes after achieving an elevation of 1 m above ground level. This corresponds to a travel distance of no more than 1 km in a 3 m/s wind. Travel distances would be 0.25 km for a 20 µm particle and 4 km for a 5 µm particle

Figure 4-1. Example of horizontal flux measurement system around an unpaved road treated with different suppressants (Watson et al., 1996b).

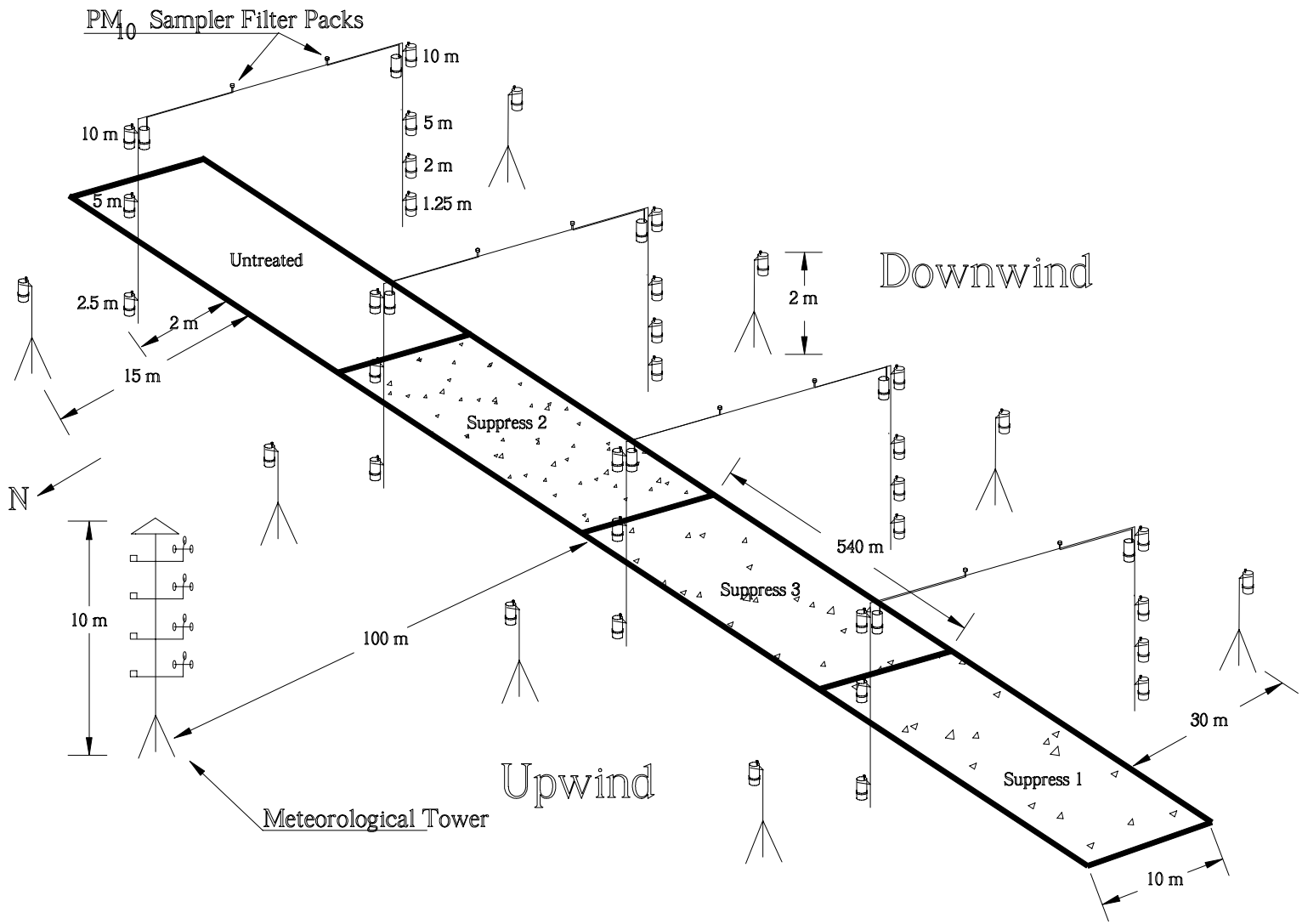


Figure 4-2. Cumulative horizontal PM_{10} flux at different downwind elevations above different unpaved roads (data from Cowherd, 1999).

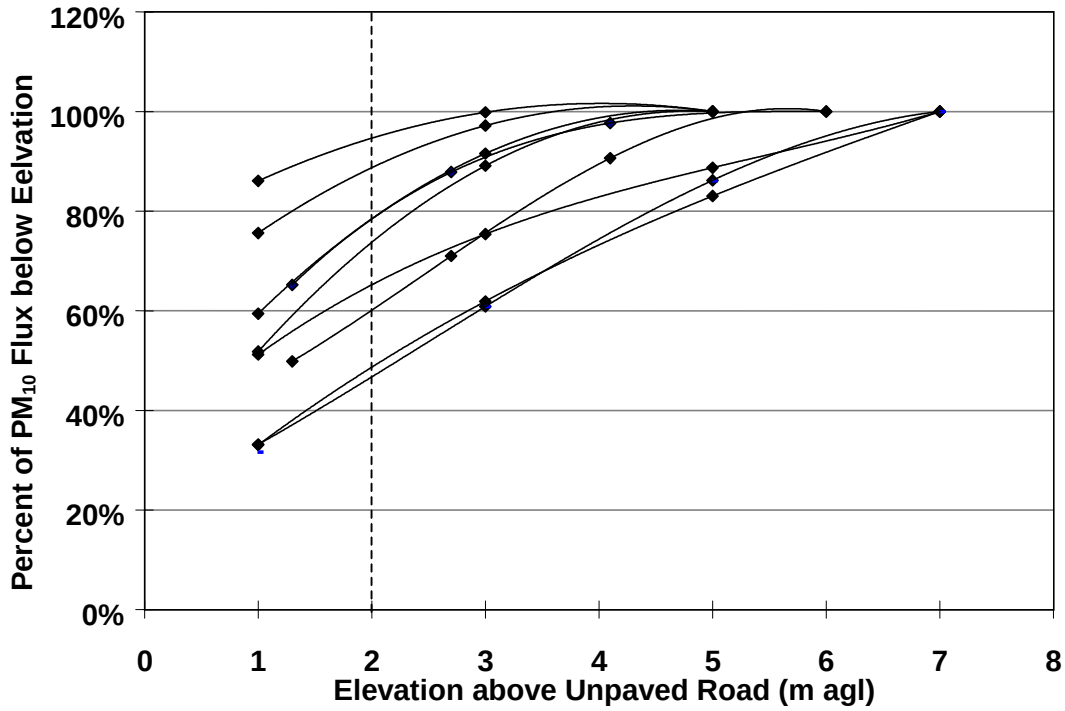


Figure 4-3. Cumulative horizontal PM_{10} flux at different downwind elevations above different paved roads (data from Cowherd, 1999).

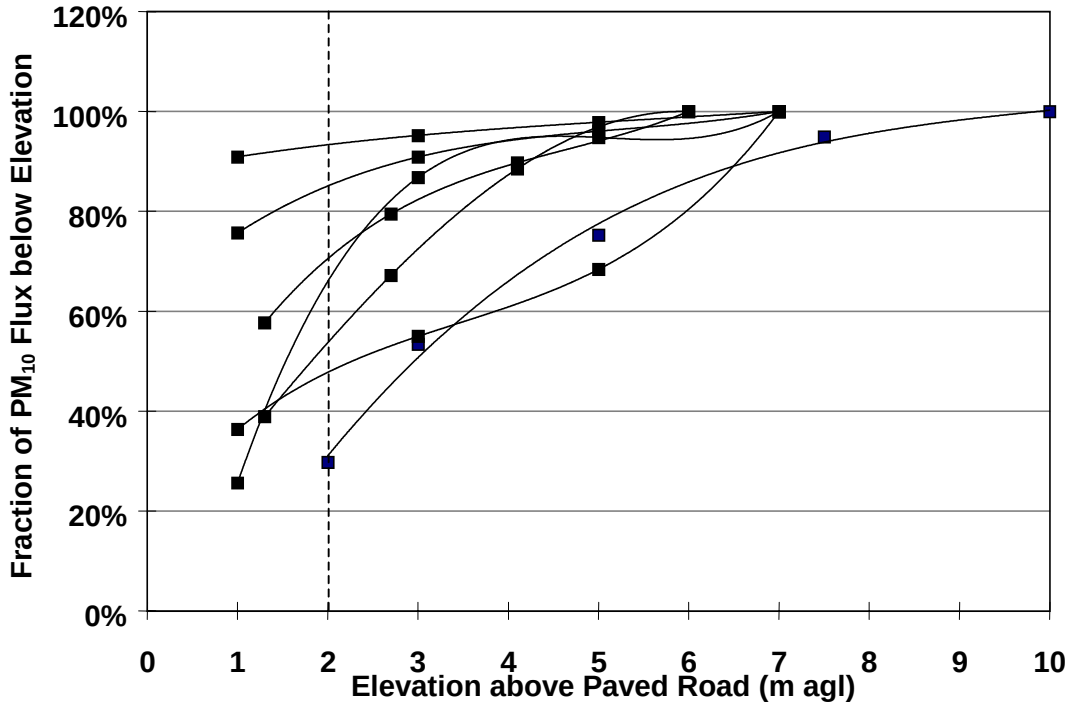
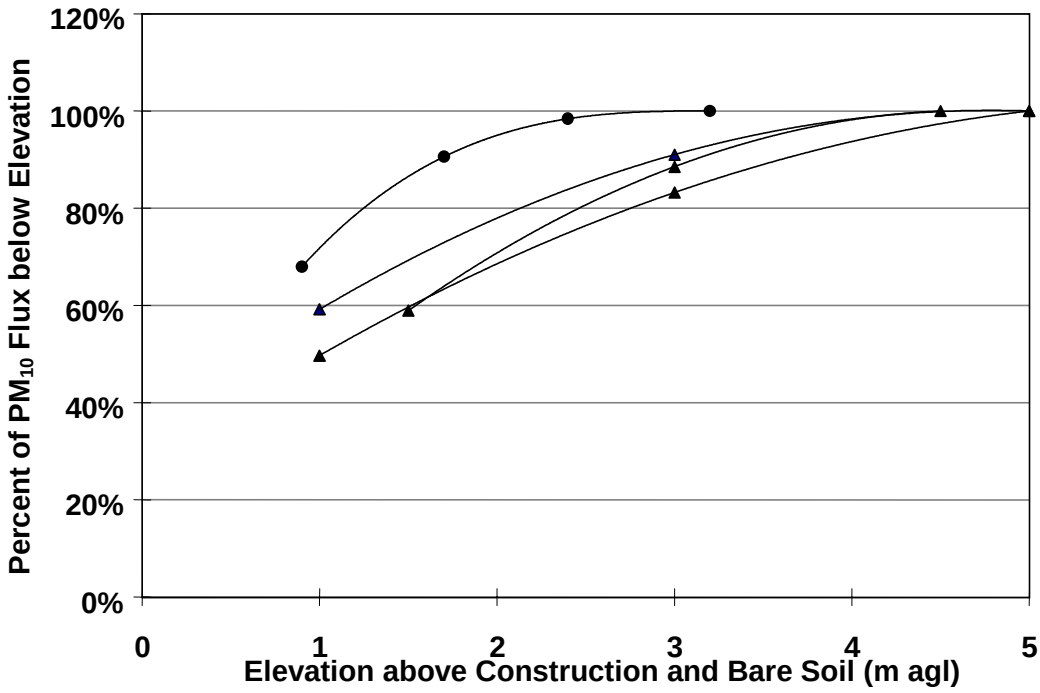


Figure 4-4. Cumulative horizontal PM_{10} flux at different downwind elevations above different construction and bare soil sites. (data from Cowherd, 1999).



under similar wind conditions. Plate-like particles (e.g., clay) fall more slowly than spherical particles. Upon returning to the surface, supermicron-sized particles can "bounce" and/or become resuspended (Chamberlain, 1967; Wu et al., 1992; Paw U, 1992). During wind erosion processes, airborne particles become electrically charged relative to surfaces, and can be attracted to those surfaces (Schmidt et al., 1998). A fairly well known manifestation of this charge separation is electrical discharges ("lightning") near the surface during some dust storms. The resulting electrical forces can substantially increase the rate of dry deposition of the airborne particles, a process apparently not currently included in dry deposition models.

If fugitive dust can be lofted from the near-surface atmospheric layer by mechanical and/or thermally generated turbulence, then particles have potential for residing in the atmosphere for substantial time periods. If the only removal process is by dry deposition, with deposition velocity v_d, and if the particles are mixed fairly uniformly to a height h_d (e.g., the height of the atmospheric mixing layer), then the residence time is essentially h_d/v_d. For example, if h_d = 1 km and if v_d = 1 cm/s, then this residence time is 10^5 sec or approximately a day.

Low-level particles are likely to deposit to the ground, horizontally impact on nearby obstructions, or rapidly disperse within a short distance from the point of emissions. The rapid attenuation of PM_{10} concentrations downwind of an unpaved road is illustrated in

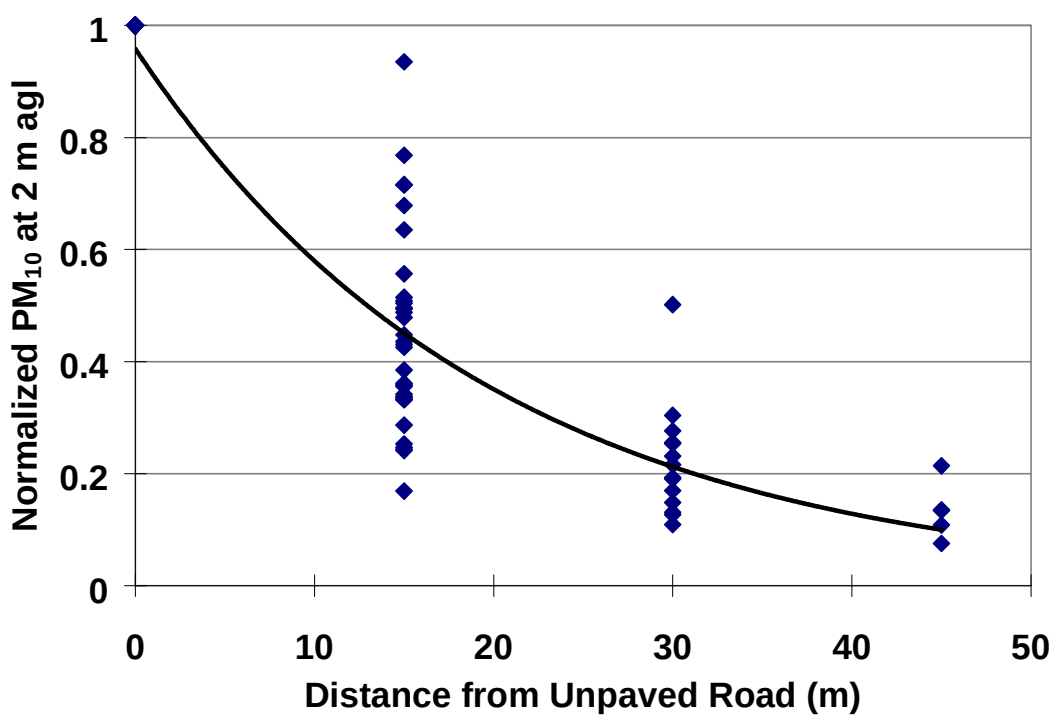
Figure 4-5. While there is substantial scatter in the ratios of downwind to roadside concentrations, it is clear that PM_{10} is attenuated by ~90% within only 50 m from an unpaved roadside.

Urban- and regional-scale dispersion models usually consider fugitive dust emissions at elevations much higher than 2 m agl. Effective emission heights of 5 to 10 m agl are often used so that a plume can develop downwind of a site. Most models are unable to resolve terrain variations of 1 to 2 m and do not account for the effects of horizontal impaction on vegetation and obstructions such as trees, shrubbery, and buildings.

There is little published information on the quantitative effect of nearby obstructions on low-level dust emissions, but it is common practice to place obstructions such as greenbelts (Kapoor and Gupta, 1984; Gupta and Kapoor, 1992) near visible dust emitters. For example, Slinn (1982) shows that dust deposition velocities through a eucalyptus forest can be five to ten times higher than gravitational settling velocities with wind speeds of 5 to 10 m/s.

Vertical flux is an alternative to horizontal flux as a method to estimate fugitive dust emissions (Gillette, 1977). Only a fraction of the horizontal flux moves upward, with this fraction dependent on a variety of meteorological, aerosol, and surface conditions. Vertical

Figure 4-5. Attenuation of PM_{10} concentrations with distance from an unpaved road (Watson et al., 1996b).



flux emissions can become available for transport to distances beyond the immediate vicinity of the source. Vertical flux is proportional to particle density, surface friction velocity, and the difference between particle concentrations at different elevations above ground level. These variables can be estimated from the same measurements used for horizontal flux measurements with a configuration similar to that of Figure 4-1 (Nickling and Houser, 1999).

It appears that currently used emission factors based on integrated horizontal flux adequately represent suspendable dust, the amount that leaves a dust-generating surface, but they do not adequately represent transportable dust, the fraction of suspendable dust that is likely to travel more than a few kilometers from the emitter.

More realistic estimates for the vertical components of the dust fluxes from fugitive emissions are needed to evaluate fugitive-dust concentrations beyond the immediate vicinity of sources (e.g., Johnson, 1983; Shao and Leslie, 1998). Shao and Leslie (1998) set the vertical flux for each particle-size range proportional to the horizontal flux, with the proportionality "constant" dependent on particle size. There is no dependence of the vertical flux on atmospheric stability.

Modeling of vertical fluxes of particles from fugitive emissions should proceed on two fronts. For regional- to continental-scale models, algorithms should be developed and tested to estimate a representative vertical flux (e.g., at the lowest layer of the air quality model, which is typically 30 m) for emissions in each grid square. This algorithm could incorporate a relatively simple boundary-layer formulation driven by the larger model's regional-scale meteorological fields. For practical applications, simple plume models that incorporate deposition and atmospheric stability, especially for high-wind-speed cases, need to be interfaced with vertical emission flux estimates.

4.1.3 Dynamic Activity Models

Dynamic activity models refer to computerized systems that have access to databases that may change over periods of months, weeks, days, or even hours. Such models could be assembled after the fact by compiling information specific to modeled episodes. They could also be created to access certain databases in real time so that the information could be used to modify activities to minimize fugitive dust emissions from certain activities such as construction. The structure of dynamic activity models can range from simple spreadsheets to complex physical interaction models that include meteorological processes to determine windflows, turbulence, humidity, and other factors that can influence fugitive dust emissions.

Spreadsheets are the easiest and most common means for constructing dynamic activity models (Kuhns et al., 1998). This approach is most appropriate to a relatively small area where the number of separate activity databases is limited and where locations can be precisely defined. Each cell in the spreadsheet can represent a grid, different worksheets can represent different activity databases, and relationships between activity and emission rates can be established by calculations in linked worksheets. The spreadsheet approach is most appropriate when used in rollback calculations, where area-wide, rather than spatially resolved, emissions are needed.

Where complex spatial emission inputs are needed, a Geographical Information System (GIS) provides the most flexible framework. Many of the available activity databases (e.g., roadway networks, land use, soil characteristics, traffic volumes) have irregular shapes onto which an arbitrary grid system must be imposed. This grid system may change for different applications. For the most part, existing spatial and temporal data sets compiled for purposes unrelated to air quality are obtained to estimate dust-generating activity.

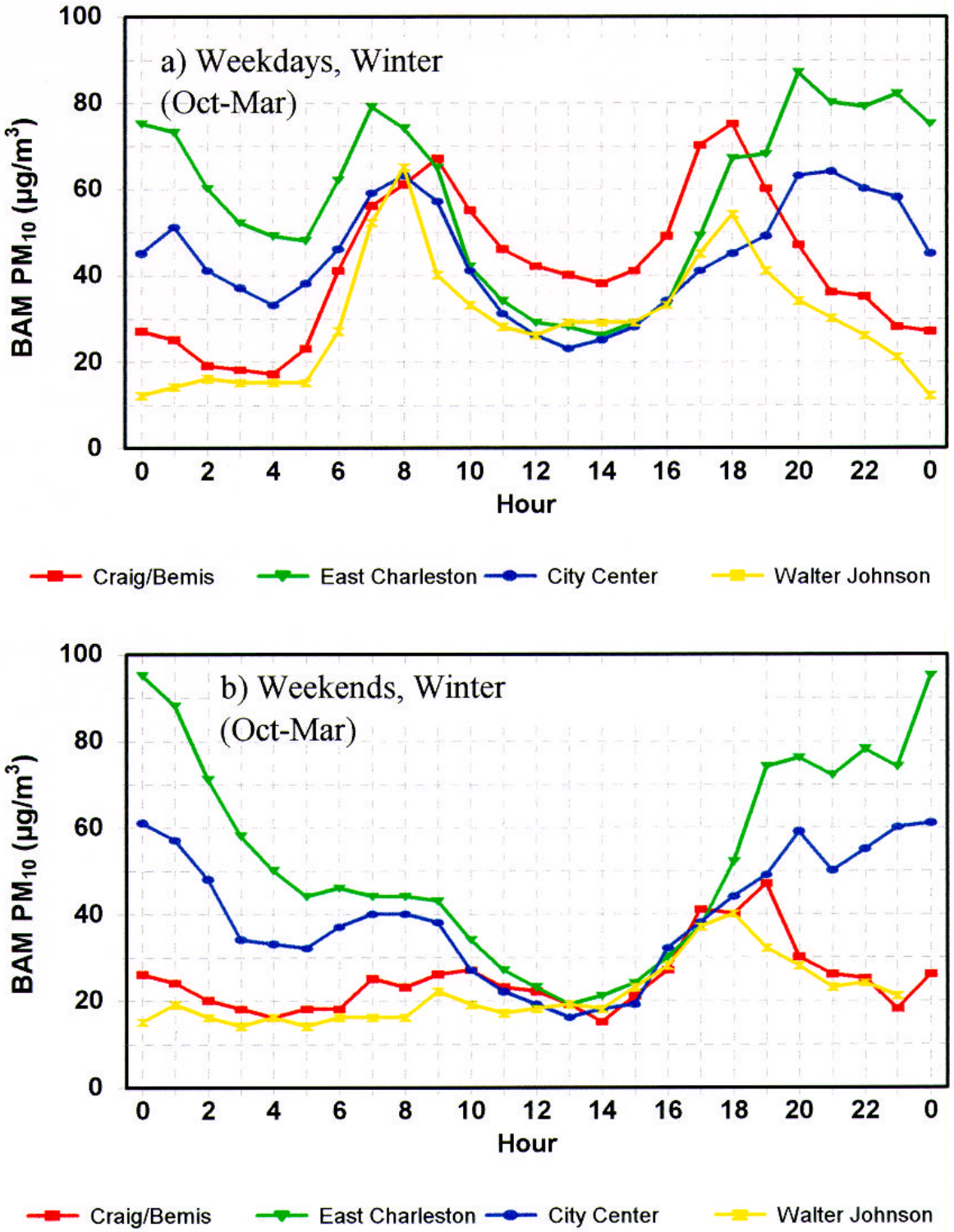
While spatial activity data sets may be obtained with seasonal, and possibly even monthly variations, it is unlikely that more highly resolved information will be available. In this case, temporal profiles can be imposed on an average emission rate for each source type. These profiles can be determined theoretically or empirically and should have diurnal as well as day-of-the-week variability. Figure 4-6 shows an example of diurnal variability in PM_{10} concentrations at different sampling sites near major roadways in Las Vegas, NV, on wintertime weekdays and weekends. Although non-traffic sources influence these concentrations, the weekday morning and evening rush hour contributions from vehicle exhaust and paved road dust are clearly discernable in the top panel. The bottom panel shows a distinctly different diurnal profile on weekends. Traffic profiles can also be developed by vehicle counting and by on-road remote sensing methods.

4.2 Dispersion Models

Source-oriented dispersion models use the outputs from emission, meteorological, and chemical models to estimate concentrations measured at receptors. Source models include mathematical simulations of transport, dispersion, vertical mixing, deposition, and chemical mechanisms to represent transformation. The most common source dispersion models are Gaussian plume, puff, and grid formulations. Gaussian plume models (Schulze, 1990; Freeman et al., 1986; Schwede and Paumier, 1997) are most often associated with the straight line wind model and estimate a bell-shaped concentration field in the vertical and horizontal directions from the wind direction. These models are commonly used to evaluate potential effects of primary emissions from ducted sources, such as industrial stacks. Puff or trajectory models treat emissions from a variety of sources as independent entities that are moved in a curvilinear wind field generated by a diagnostic or prognostic wind model. Grid models transfer pollutants between boxes with pre-defined vertical and horizontal dimensions (Bowman et al., 1995; Byun and Ching, 1999; Byun and Dennis, 1995; Yamartino et al., 1992). Current dispersion modeling of particulate matter emissions is generally accomplished using gridded Eulerian chemical-transport models (CTM's) on urban to regional scales. Smaller scales are often modeled by less complex Gaussian plume or trajectory models.

Meteorological models consist of straight line, interpolation (termed diagnostic), and first principle (termed prognostic) formulations, with increasing levels of complexity and requirements for computational and data resources. The straight line model is applied to hourly wind directions from a single monitor, assuming an air mass travels a distance equal to the wind velocity in the measured direction, regardless of the distance from the monitoring

Figure 4-6. Hourly variations in PM_{10} concentrations acquired by beta attenuation monitors (BAM) at different urban residential (Walter Johnson), commercial (East Charleston, City Center) and industrial (Craig/Bemis) sites in Las Vegas, NV on: a) weekdays, and b) weekends (Chow and Watson, 1997).



site. This model is applicable for a few hours of transport in flat terrain, typically for evaluating a single emission source. Interpolation models integrate wind speed and direction data from multiple measurement locations, including upper air measurements provide by remote sensors or balloon launches. The more advanced of these models allow barriers, such as mountains, to be placed between monitors. Wind fields, therefore, show different directions and velocities at different horizontal and vertical positions. Interpolation wind models are applicable to domains with a large number of well-placed monitors and for estimating the movement of air masses from many sources over transport times of more than half a day. The number and placement of monitors, especially upper air monitors, is especially important in mountainous terrain and in coastal areas where winds are unusual.

Advanced chemical transport models are driven with meteorological data taken from prognostic models (Stauffer and Seaman, 1994; Seaman et al., 1995; Koracin and Enger, 1994) that embody scientists' best knowledge of atmospheric physics and thermodynamics. These models employ basic equations for conservation and transfer of energy and momentum. These meteorological models are computationally intensive, often requiring supercomputers but are becoming more practical and cost-effective as workstation and desktop computers become more powerful. Modern versions use "four-dimensional data assimilation" that compare model-calculated wind, humidity, and temperature fields with measurements and "nudge" model outputs toward observations. A more complex meteorological model is not necessarily a better model for a specific application. The MM5 meteorological model has been adopted as the platform for several air quality studies (Seaman et al., 1995).

Emission, chemical transport, and meteorological models must be applied to the same temporal and spatial scales. Horizontal dimensions of chemical transport models range from as low as 1 to 2 km for urban-scale applications to 10 to 50 km for regional-scale applications. Vertical dimensions are treated as layers at different elevations above ground level with thickness ranging from 20 m to 2,000 m. Most of the model inputs and outputs are assumed to be homogeneously distributed within rectangular solids with these dimensions. This assumption implies that inputs and outputs with large spatial variability within these dimensions will not be well simulated by these models.

Emission data needed for regional modeling includes both emission fluxes and some characteristics of the emissions and their sources. As noted in Section 4.1, fugitive dust emissions are derived from horizontal flux estimates. Vertical emission fluxes are better simulated by chemical transport models. Emissions inventory inputs to chemical transport models include: 1) source location identification (coordinates for point sources, political identifiers for political areas); 2) pollutant identification (PM_{10}, $PM_{2.5}$, or other size fractions); 3) emission flux (mass per unit time); 4) source type; 5) pollutant emission controls (if any); 6) effectiveness of controls (if any); 7) applicable time frame of inventory; and 8) for point sources with stack height, identifier, diameter, flow rate, exhaust temperature, and exit velocity. There is large contrast between the detailed information available for industrial sources and the less complete information for area and mobile sources.

4.3 Receptor Models

The Chemical Mass Balance (CMB) air quality model is one of several models that have been applied to air resources management. As described in Section 2, receptor models use the chemical and physical characteristics of gases and particles measured at source and receptor to both identify the presence of and to quantify source contributions to receptor concentrations. Receptor models are generally contrasted with dispersion models that use pollutant emission rate estimates, meteorological transport, and chemical transformation mechanisms to estimate the contribution of each source to receptor concentrations. The two types of models are complementary, with each type having strengths that compensate for the weaknesses of the other.

The CMB consists of a solution to linear equations that express each receptor chemical concentration as a linear sum of products of source profile abundances and source contributions. The source profile abundances (i.e., the mass fraction of a chemical or other property in the emissions from each source type) and the receptor concentrations, with appropriate uncertainty estimates, serve as input data to the CMB model. In order to distinguish among source type contributions, the measured chemical and physical characteristics must be such that they are present in different proportions in different source emissions and changes in these proportions between source and receptor are negligible or can be approximated. The CMB calculates values for the contributions from each source and the uncertainties of those values.

Receptor models do not need emission rates from an emission model, but they do need chemical profiles that allow one source to be distinguished from other sources. For quantitative source apportionment, source profile properties must meet the following criteria: 1) their abundances (mass fraction) are different in different source types; 2) their abundances do not change appreciably during transport between source and receptor (or such changes can be simulated by measurement or modeling); and 3) their abundances are reasonably constant among different emitters and operating conditions for a selected source type. These profiles are used in emission inventories to separate mass emissions into chemical component emissions and in receptor models to apportion ambient concentrations of suspended particles to sources.

Additional chemical and physical properties beyond the commonly measured elements, ions, and carbon need to be examined to determine which ones can meet these criteria in practical source apportionment applications. Several types of analyses that might meet these needs are:

- **Appearance:** Optical microscopy provides an overview, albeit semi-quantitative, of the shape, size, mineralogy, and type of different particles with geometric diameters larger than ~2 μm.

- **Soluble and insoluble inorganic elements and compounds:** X-ray fluorescence, proton-induced x-ray emission, instrumental neutron activation analyses, atomic absorption spectrophotometry, inductively coupled plasma atomic emission spectroscopy, x-ray diffraction, computer automated scanning

electron microscopy, ion chromatography, and automated colorimetry have been applied, sometimes in conjunction with different extraction and selective sample preparation techniques, to quantify a variety of elements and crystalline structures.

- **Carbon groupings:** Organic and elemental carbon, thermal desorption and pyrolysis patterns, and solubility in different organic solvents using thermal manganese oxidation, thermal optical reflectance, thermal optical transmittance, Fourier transform infrared spectroscopy, thermal desorption gas chromatography and mass spectrometry, and sequential solvent extraction are possibilities.

- **Specific organic compounds:** Pesticides, herbicides, cellulose, and many other specific organic compounds may be analyzed by solvent extraction followed by gas or liquid chromatography with different detectors.

- **DNA, toxins, microbes, and bacteria:** Biologically specific tests can be applied to characterize these substances.

- **Isotopes:** Low-level radioactive counting, dilution mass spectrometry, and accelerator mass spectrometry are potential methods to quantify unique isotopic abundances in geological materials.

A systematic survey of the extent to which these observables are found in different sources is needed to further specify source attribution using receptor models.

4.4 Interaction Among Models

Given the dynamic nature of PM emissions, more accurate representations may be gained by modeling the emissions as described in Section 4.1. Emission modeling needs to bring emissions to the same temporal and spatial scales, typically 1 to 10 km and 1 hour resolutions, that are used for the meteorological and chemical/transport models.

In current practice, emissions are considered static and do not interact with the modeling process. A better approach would apply a dynamic emission model that uses meteorology and changing activity databases as inputs. This dynamic emission model would account for the transportable fraction of dust emissions that is consistent with the spatial scales of the other models. For microscale modeling of neighborhood impacts, this might include horizontal fluxes in the lowest levels. For urban and regional modeling with 1 to 10 km spatial domains, only horizontal fluxes at higher elevations, or vertical fluxes, would be used to represent transportable particles.

Receptor models such as the CMB should be included in this interaction. The first role of receptor models is the direct apportionment of ambient concentrations to their sources. This apportionment in and of itself has been of great utility in developing control strategies in PM_{10} non-attainment areas. The second role of receptor models is to independently verify emissions inventories, as in the comparison between inventory and

source apportionment studies presented in Section 2. Impediments to CMB/inventory reconciliation are the lack of specificity of chemical components that distinguish different fugitive dust source types from each other. Receptor models also offer a framework for evaluating the outputs of complicated air quality models. Source-oriented models produce a pattern of concentrations and uncertainties similar to CMB source profiles that could be used by the CMB for a multivariate fit to the ambient data. This is a more complex, and potentially more accurate "profile aging" process, as represented in a simpler context by Friedlander (1981).

4.5 Hypothesis Testing

The preceding discussion allows the following conclusions to be drawn about hypotheses five and six:

- **Insufficient and uncertain activity levels.** Current emissions inventories are not dynamic. They do not easily adapt to the changing times and locations of activities that affect ambient concentrations. Dynamic emission models are needed that seamlessly interact with meteorological, chemical/transport, and receptor models. Chemical source profiles need to be enhanced with more specific markers for different fugitive dust source types to verify changes in activity through receptor measurements. Dynamic databases of the locations and timing of emissions need to be incorporated into emission estimation methods.

- **Insufficient accounting for injection heights, deposition losses, and horizontal impaction losses in dispersion models.** This is the most evident and documented cause of discrepancies between emission estimates and ambient source contributions from fugitive dust. Commonly used fugitive dust emission factors estimate integrated horizontal fluxes from ~1 m to ~10 m above ground level. Approximately 60% to 90% of these fluxes are below 2 m and will not travel long distances (> a few km) from the emitter. Most receptor model source contributions are estimated at community representative, urban-scale, monitoring sites that are specifically selected to be uninfluenced by nearby sources. Most source-oriented models estimate source contributions averaged over dimensions of 1 km to 10 km. Transportable dust emissions need to be distinguished from suspendable dust that is currently used to estimate emissions. Transportable dust emissions depend on the spatial scale of modeling. For middle-scale models, current estimates are probably adequate. For neighborhood-, urban-, and regional-scale modeling, the transportable fraction is likely to vary. Methods to estimate this variability need to incorporate vertical fluxes and removal processes due to vertical deposition and horizontal impaction. A first approximation for transportable dust that moves beyond a few kilometers would eliminate the horizontal flux within the first 2 m above ground level. This would reduce current fugitive dust emission estimates by ~75%.

5. SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

Previous sections of this report identified differences between emission estimates and ambient geological contributions, estimated the magnitudes of these differences that could be caused for different reasons, and evaluated the technical bases for currently applied emission estimation procedures. This section summarizes the results of this examination and specifies research projects that can be conducted immediately, in the short term (1 to 2 years), and over a longer term (2 to 5 years) to improve scientific understanding of and quantitative estimation of fugitive dust emissions.

One of the methods used to identify potential discrepancies was to compare emissions estimated for a single state, California, with national inventory estimates for that state. The methodology is essentially the same for these inventories, but the emission factors, activity databases, and control measures applied are substantially different. These differences resulted in California's urban dust emissions from unpaved roads, paved roads, and construction being only 60% of those estimated by the national inventory. Much larger discrepancies were found for specific areas and source types.

5.1 Summary

Subtracting secondary sulfate and nitrate contributions from PM_{10} prior to estimating the fugitive dust fraction in source apportionment studies results in a higher fractional geological contribution. There is still a large discrepancy between the dust fraction in emissions inventories and the fractional dust contribution to ambient mass concentrations. Although $PM_{2.5}$ source apportionment studies are limited, it is likely that discrepancies will be larger for this size fraction in which geological contributions are much lower. A more systematic examination of the fugitive dust proportion in ambient samples will be possible when data become available from EPA's $PM_{2.5}$ speciation network at representative community exposure sites throughout the U.S.

Spatial distributions of annual average $PM_{2.5}$ soil contributions range from 0.5 to 1 $\mu g/m^3$ at background sites in the United States as determined by the Interagency Monitoring of PRotected Visual Environments (IMPROVE) network. These should be subtracted from urban PM_{10} and $PM_{2.5}$ fugitive dust contributions prior to determining proportions for comparison with local emissions inventories. Higher soil contributions on a few days in Florida and other areas may be caused by long-range transport of Sahara dust that are not accounted for in inventories. More extensive IMPROVE monitoring in the eastern U.S. will supplement urban speciated $PM_{2.5}$ monitoring sites to improve estimates of background contributions to urban fugitive dust levels. More specific chemical and physical markers need to be developed and measured in source and receptor samples to distinguish background from other fugitive dust sources, as well as different types of dust emitters from each other.

Temporal and spatial averaging periods for fugitive dust emissions are not the same as those for ambient $PM_{2.5}$ and PM_{10} measurements. National inventories are specific to statewide averages and are based on activities averaged over one year or more. State and local inventories rarely have spatial resolutions better than the size of a county and temporal resolution of less than a month. These averaging times contrast with intermittent and often

random events that raise dust into the atmosphere to affect ambient samples that are averaged over 24-hours or less. Discrepancies between inventory and ambient concentrations are especially evident in large counties that contain population centers with PM_{10} or $PM_{2.5}$ monitors. A comparison between California and national emissions for unpaved roads, paved roads, and construction at the county level showed substantial differences by county and by source category, even though the statewide totals were 40% lower than the national totals. These are probably due to differences in the statewide activities that are allocated to county in the national inventory compared to the countywide activities that are aggregated to state in California. Most of the emissions from unpaved roads and windblown dust occur in non-urban areas far from the monitor, but they are included in the inventory used to assess source contributions at that monitor. Even within an urban area, neighborhood-scale studies show that there may be large differences between dust contributions for monitors separated by a few kilometers. To improve the relationship between emissions and source contributions, the spatial and temporal resolutions of emissions inventories need to be made more compatible with the zone of representation and averaging times of PM monitors.

Emission factors for unpaved road and paved road dust emissions are reasonably consistent from one test to another. Major reductions were found for California when location-specific values were used for silt content and loadings in place of the default values used for the national inventory. Construction emission factors used in the national inventory overestimate construction emissions because they assume that the heavy earthmoving activities on which they are based occur throughout the duration of the project. California inventories use construction emission factors derived from the actual processes that engender those emissions and are an order of magnitude lower than those used for the national inventory.

National inventory activity levels do not accurately reflect the activities at county or finer resolutions. National estimates are made statewide, then allocated to counties based on activity surrogates. There are clear discrepancies between this approach and the California inventory that builds the statewide inventory from countywide activity estimates. The county specific data better allocate emissions than the statewide national activity databases. On the other hand, the national inventory assumes control effectiveness values that are not borne out by fugitive dust control demonstration studies. There are still large uncertainties and variabilities in the activity databases for both inventories.

Current emissions inventories are not dynamic. They do not easily adapt to the changing times and locations of activities that affect ambient concentrations. Dynamic emission models are needed that seamlessly interact with meteorological, chemical/transport, and receptor models. Chemical source profiles need to be enhanced with more specific markers for different fugitive dust source types to verify changes in activity through receptor measurements. Dynamic databases of the locations and timing of emissions need to be incorporated into emission estimation methods. The wind erosion portion of the national inventory represents a relatively small number of individual events that are averaged over a year. These events should not be included in annual average estimates that are intended to represent continuous and repetitive processes.

There is insufficient accounting for injection heights, deposition losses, and horizontal impaction losses in dispersion models. The scale and temporal resolution of emissions inventories is incompatible with the needs of dispersion models. This is the most evident and documented cause of discrepancies between emission estimates and ambient source contributions from fugitive dust. Commonly used fugitive dust emission factors estimate integrated horizontal fluxes from ~1 m to ~10 m above ground level. Approximately 60% to 90% of these fluxes are below 2 m and will not travel long distances (> a few km) from the emitter. Most receptor model source contributions are estimated at community representative monitoring sites that are located for minimal influence from nearby sources. Most source-oriented models estimate source contributions averaged over dimensions of 1 km to 10 km. Transportable dust emissions need to be distinguished from suspendable dust that is currently used to estimate emissions. Transportable dust emissions depend on the spatial scale of modeling. For middle-scale models, current estimates are probably adequate. For neighborhood-, urban-, and regional-scale modeling, the transportable fraction is likely to vary. Methods to estimate this variability need to incorporate vertical fluxes and removal processes due to vertical deposition and horizontal impaction. A first approximation for regionally transportable dust (i.e., dust that is likely to travel more than a few kilometers) could assume that horizontal flux within the first 2 m above ground level is removed by nearby surfaces. This would reduce current fugitive dust emission estimates by about 75%.

5.2 Conclusions

- National inventories do not accurately estimate emissions from fugitive dust sources that affect ambient $PM_{2.5}$ and PM_{10} concentrations for a variety of reasons. Both negative and positive biases in emissions result, but the net effect is to overestimate emissions from fugitive dust sources relative to those from other sources in the inventory.

- Suspendable particles are not transportable particles. Available data shows that ~75% (ranging from ~60% to ~90%) of suspended PM_{10} remains within 1 to 2 m above ground level. These particles deposit to the surface or impact on nearby vertical structures within a few minutes after suspension. Although horizontal fluxes commonly used in empirically-derived emission factors represent the mass of dust particles suspended from a surface, they do not represent the mass entrained into the atmosphere and transported over distances of more than a few kilometers.

- Source and receptor models do not represent the same spatial and temporal scales as emissions inventories. Better integration with and interaction among emission, meteorological, chemical/transport, and receptor models is needed. This interaction should help to identify and reconcile deficiencies in all of the modeling components.

- State and local values for emission factors and activities yield lower emission estimates, at least for California. Several of these emission factors and activities should be used in national inventories.

- Emissions inventories treat fugitive dust emissions as continuous processes, whereas they are intermittent processes that depend on many meteorological and activity variables. This causes a positive bias in dust emission estimates.

- Few empirical tests are available for the $PM_{2.5}$ fraction of fugitive dust emission factors. The extent to which TSP and PM_{10} emission factors can be scaled to $PM_{2.5}$ is uncertain.

- Paved and unpaved road dust emission factors have been found to be of similar form and magnitude for many independent tests for TSP and PM_{10}. Silt loadings, silt fractions, and activity levels used in national inventories appear to overestimate emissions from these sources, at least with respect to California.

- Existing tests for emission factors are biased toward the highest emitters, yet they are applied to a population of activities that encompasses a wide range of emission magnitudes. Lower-emitting representatives of a source category are poorly characterized. This is especially true for construction emission estimates that characterize the earth moving portion of a project and not the construction activities.

5.3 Immediate Recommendations for National Emissions Inventory Improvement

- Review construction emission factors and compare them to those used in California. Consider adopting similar factors for the national inventory.

- Use measurements from previous fugitive dust emission tests to estimate horizontal dust fluxes above elevations of 2 m. Create separate estimates for emissions above these heights that can be used for different modeling and planning purposes. Urban- and regional-scale source or receptor modeling could use the >2 m above ground level horizontal fluxes. Preliminary estimates given in this report indicate that >2 m horizontal fluxes are ~25% of total horizontal fluxes. This approximation needs more scientific justification.

- Conduct countywide comparisons between national inventory and independently compiled statewide inventories. Use the results of these comparisons to identify the availability of or need for local information that affect suspendable dust surface loadings and dust-generating activities.

5.4 Short-Term Recommendations for National Emissions Inventory Improvement

- Create and apply methods to estimate vertical flux, as well as horizontal flux, from fugitive dust sources and add vertical flux estimates to modeling inventories.

Revise the current emission measurement methodology such that vertical fluxes and horizontal fluxes above selected elevations are incorporated into emission factors and are reported in inventories.

- Conduct additional emission tests of unpaved roads, paved roads, construction, and other earth-moving emissions that apply vertical and horizontal flux methods and are specific to $PM_{2.5}$ and PM_{10} particle sizes. These tests should represent a variety of areas in the eastern and western United States.

- Conduct detailed studies of temporal variability for underlying activities (including control measure effectiveness) that create dust. Determine representative temporal profiles for diurnal, weekly, monthly, and annual emissions for each source type. Determine the extent to which reservoirs of dust are depleted and incorporate these into emission models.

- Create a modeling framework that integrates emission, meteorological, chemical/transport, and receptor models. This framework would use the meteorological model to estimate wind speed, wind direction, and moisture that affect fugitive dust emission rates. Receptor models applied to ambient data would be used to estimate source contributions at receptors and to reconcile emissions with these contributions on an area-specific basis.

5.5 Long-Term Recommendations

- Modify existing air quality modeling software to better represent the vertical flux, deposition, and transport of dust for different spatial scales. Develop, test, and apply more realistic middle- and neighborhood-scale mathematical models to represent dust concentrations at receptors near a variety of dust emitters.

- Develop a complete, GIS-based, emission model that can be applied to neighborhood, urban, and regional scales. This model would include and update commonly available activity databases such as land use, roadways, and soil surveys. It would provide for the use of temporal profiles that allow for daily, weekly, monthly, and annual emission estimation. The model would also contain fugitive dust source profiles suitable for receptor model source apportionment and for creating speciated inventories that can be used for rollback modeling. It should propagate input data uncertainties, project emissions under different development scenarios, and allow for alternative emission factors and activity databases to estimate the same emission rates.

- Compile dust characteristics, including surface loadings, chemical compositions (source profiles), and suspendable particle content (silt or other) for representative locations throughout the U.S. Include these in a documented database and apply them to development of emission estimates. Priorities should be given to airsheds that operate $PM_{2.5}$ chemical speciation monitors. A protocol that combines systematic grab sampling and laboratory analysis with continuous on-road measurements is needed for this to be practical.

- Calculate source contributions with receptor models applied to data from the national $PM_{2.5}$ speciation network. Reconcile these contributions with fugitive dust emission estimates from these areas and with background concentrations from nearby IMPROVE monitoring sites. Use the results of this reconciliation to further identify and ameliorate emission modeling deficiencies.

- Identify and characterize chemical or physical components in different fugitive dust sources that allow fugitive dust sub-types to be distinguished from each other in receptor samples. Apply these characterization methods to receptor samples, quantify contributions at representative distances from sub-type emitters, and use the results to improve fugitive dust emission estimation methods.

- Define and conduct experiments that increase understanding of vertical flux, deposition, and removal by surrounding barriers. Such experiments might include eddy-correlation micrometeorological measurements of positive and negative vertical fluxes, balances of airflow through and over tree stands downwind of dust emitters, and measures of particle rebound and filtration over short vegetation, such as grass.

- Develop and apply new methods to estimate fugitive dust emissions using real-time and remote sensing methods. These methods would be used to verify emissions from the more established and commonly applied methods as well as to better understand the physical interactions between emissions and the atmosphere.

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