METHODOLOGY FOR ESTIMATING FUGITIVE WINDBLOWN AND MECHANICALLY RESUSPENDED ROAD DUST EMISSIONS APPLICABLE FOR REGIONAL SCALE AIR QUALITY MODELING

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EXECUTIVE SUMMARY

The Grand Canyon Visibility Transport Commission (GCVTC) was created in response to the Clean Air Act of 1990 with the goal of identifying measures that could be implemented to reduce emissions and improve visibility in the Colorado Plateau. The GCVTC prepared a report for the EPA in 1995 that included an emission inventory for the study region, outlined several potential control measures and identified areas of investigation to be pursued in the future. One of the recommendations was to investigate the near-field and far-field effects on visibility of mechanically resuspended fugitive dust from paved and unpaved roads in Class I areas in the intermountain west. Several years later, near source removal of fugitive dust particles was one of the primary issues addressed at a EPA sponsored workshop convened to identify research needs to reconcile fugitive dust emissions inventories with estimates of ambient source contributions for urban areas.

In 1997 a successor organization to the GCVTC was formed: the Western Regional Air Partnership (WRAP). The Research and Development (R&D) Forum of the WRAP, with the responsibility of advising the WRAP on technical issues, identified windblown fugitive dust as an area needing further study in addition to mechanically resuspended road dust. In 2000 a panel of air quality experts was convened by the R&D Forum to identify the best methodology currently available for estimating emissions of fugitive windblown and mechanically resuspended road dust applicable for regional scale air quality modeling, and to make recommendations for future research activities to generate improved fugitive dust emissions estimation techniques applicable for regional scale modeling. This panel derived a list of recommendations based on eleven findings regarding fugitive windblown and mechanically resuspended road dust.

Expert Panel Findings and Recommendations

<u>Finding #1: Not all suspendable particles are transported long distances</u> Only a fraction of suspendable particles are regionally transportable particles. Ground level emissions of mechanically generated particles are likely to be removed near the source due to gravitational settling as well as impaction on nearby obstacles, with large particles having a greater removal rate than small particles. The initial vertical energy associated with mechanically generated particles is typically short-lived and unsustained. In the absence of violent winds with large vertical components (such as those in dust devils or thunderstorms) or significant solar heating of the ground to cause upward diffusion due to large turbulent eddies, there is little, if any, residual or continuing energy to sustain vertical motion and transport of these emissions away from the source. For winds accompanied by gusty conditions or high turbulence, windblown dust emissions may be lofted vertically to great heights above the ground by the sustained energy provided by the vertical component of the wind and transported long distances from the source.

<u>Recommendations based on Finding #1</u>
(1.1) Conduct PM_{10} and $PM_{2..5}$ upwind/downwind experiments at different elevations around roadways and exposed surfaces similar to those used to develop TSP and PM_{10} horizontal flux measurements in order to determine the flux of particles at different heights.

(1.2) Determine values for relevant parameters (e.g., barrier height, length, and permeability, as well as surface roughness and friction velocity) for different ground covers for different seasons. These parameters should then be used in emissions models to estimate how wind speeds are attenuated and to derive accurate estimates of deposition velocities that remove particles from long-range transport.
(1.3) Make preliminary estimates of deposition velocity for these parameters and link to a gridded database of land cover across the Western US.

Finding #2: Regional scale vertical flux is smaller than the local-scale fugitive dust flux The regional scale vertical dust flux for a large scale transport model applies only to particles that are not deposited in the same grid cell from which they are emitted. Since a portion of the particles are deposited within the same grid cell from which they are emitted, the effective regional scale vertical flux of fugitive dust particles is smaller than the local scale vertical flux. The ratio of the effective regional scale vertical flux to the local scale vertical flux is a function of the friction velocity of the surface and the deposition velocity of the different size particles.

Recommendations based on Finding #2
(2.1) Test the validity of Gillette's semiempirical box model in a relatively clean environment where fugitive dust from vehicular traffic is a dominant source of PM_{10}.
(2.2) Upgrade the model with more complex and representative submodels to account for deposition and diffusion near the surface.

Finding #3: Fugitive dust emission factors need to be appropriate Many of the past fugitive dust emission inventories were inaccurate since they used inappropriate emission factors. Emission factors should be based on physical models rather than statistically significant variables and should be consistent for different source types with similar suspension mechanisms. There have been recent advancements in characterizing the factors that make up the empirically derived emission factor equations as well as the reformulation of the emission factor equations that need to be taken into account. For example, improved algorithms for fugitive dust emissions from construction operations are currently being implemented. Also, the unpaved road dust algorithm is currently under review. Some categories may still be inaccurate because re-evaluations have not been undertaken. Categories needing more work are dust from unpaved roads and wind erosion. There are inconsistencies in the way different regulatory agencies calculate windblown dust emissions. Scientists must agree on an approach that considers the physics of soil suspension by high winds.

Recommendations based on Finding #3
(3.1) Field test the performance of the Wind Erosion Prediction System (WEPS) model for predicting PM_{10} emissions during windblown dust episodes.
(3.2) Adopt the less data intensive modeling approach of Draxler et al. (2000) for predicting PM_{10} emissions during windblown dust episodes. This method requires a limited amount of *a priori* surface information to estimate the threshold friction velocity and calculate the horizontal flux. The proportionality constant for estimating windblown dust emissions from the horizontal flux will be needed for specific soils. As a starting point, an examination of archived data sets that include both horizontal soil fluxes and vertical dust fluxes will provide values of this proportionality constant for various soils.

(3.3) Evaluate alternative forms for EPA's empirical emission factor equations listed in AP-42, and revise these equations in keeping with a mechanistically based physical model in order to improve the accuracy of emissions predictions. This evaluation should include reviewing recent emission factor development (exposure profiling) studies. In addition, the vertical concentration distribution measured during exposure profiling studies should be evaluated to determine if the data are sufficient to make an assessment of the vertical flux.
(3.4) Acquire information on silt loadings, silt content and moisture content (i.e., the "correction" factors utilized by the emission factor equations) at the county or sub-county level for areas upwind of Class I areas for different periods throughout the year. This information should be updated regularly.

Finding #4: Source activity levels need to be accurate Fugitive dust emission inventories have often used inappropriate estimates of the extent of the source activity levels. There is a large uncertainty in the extent of the reservoir of suspendable particles (which is especially true for wind erosion and paved roads), as well as the effects of meteorological variables and human intervention. Available activity databases need to be identified and evaluated with respect to utility and scale, especially for scales of local and regional source influence at western Class I areas. Emission factor development efforts should consider the availability of activity level data and produce factors normalized to the reported activities.

Recommendations based on Finding #4
(4.1) Acquire information on source activity levels at the county or sub-county level for areas upwind of Class I areas for fugitive dust emissions for both windblown dust and mechanically suspended road dust for different periods throughout the year, and update this information regularly.
(4.2) Identify the mechanisms leading to particle reservoir replacement; and quantify the time period required for replenishment and the effects of this replenishment process on emission estimates.

Finding #5: Fugitive dust emissions are not continuous processes Emission inventories (and the emission factors used to develop emission inventories) incorrectly treat fugitive dust emissions as continuous processes.

Recommendations based on Finding #5
(5.1) Investigate the use of "puff" type dispersion models, that assume emissions are instantaneous rather than continuous, with upwind/downwind exposure profiling measurements to back-calculate source strengths and develop emission factors.
(5.2) Characterize emission rates for short time frames.

Finding #6: Annual fugitive dust emission inventories are not sufficient Improved seasonal and diurnal profiles are needed for use in emission inventories since annual emission inventories are not sufficient to develop emission control strategies for haze events. Wind erosion is highly variable and poorly characterized. Many of these variations are affected by certain meteorological conditions that are not currently considered in emissions models.

Recommendations based on Finding #6

(6.1) Account for emissions on a seasonal basis, for example following the California methodology that includes identifying and evaluating agricultural, meteorological, and land use data bases for selected Class I areas.

(6.2) Adapt nephelometer sampling schedules and wind measurements at selected IMPROVE sites to 5-minute averages to better detect high concentrations of suspended particles that might arise from fugitive dust sources.

(6.3) Operate continuous particle monitors at 30-minute or shorter time resolution at selected IMPROVE sites; examine these data to determine the fraction of a 24-hour sample that is contributed by short duration events; and evaluate the magnitude of these events relative to longer-term emissions estimates.

Finding #7: Spatial allocation of fugitive dust emissions is important Better spatial surrogates than are currently in use are available for estimating fugitive dust reservoirs, locations, dust-generating activities, and temporal changes in surface properties and surroundings. Use of these spatial surrogates would provide better emissions estimates.

Recommendations based on Finding #7

(7.1) Catalog, describe, and evaluate spatial data bases such as soil surveys, digital road maps, satellite and other land use data, meteorological measurements and models to interpolate and extrapolate measurements, and traffic demand estimates; determine the availability and costs of these data; and identify technical and cost impediments for using them to improve fugitive dust inventories.

(7.2) Develop and apply a systematic program to sample representative soils; determine their PM_{10} and $PM_{2.5}$ indices based on the methodology of Carvacho et. al (1996) which uses the ASTM wet sieve method rather than the dry sieve method used by EPA (i.e., in AP-42); relate these to soil texture properties in soil surveys; and use these to estimate suspendable particle reservoirs on open lands.

(7.3) Develop and apply a practical method to obtain continuous roadway dust loadings (e.g., forward-scattering nephelometer); evaluate the potential to apply this method during normal driving cycles of park personnel and others in Class I areas; evaluate these data to determine statistical distributions of surface loadings; and determine how they change with different variables.

(7.4) Develop and apply a flexible GIS emissions modeling structure that continuously acquires and updates spatial surrogates from existing and planned data bases and propagates this new information into better estimates of reservoirs, dust-suspending activities, and attenuation caused by obstacles near the point of emission.

Finding #8: The fine fraction of fugitive dust emissions is not adequately characterized Few empirical measurements exist for the $PM_{2.5}$ fraction of fugitive dust emissions (i.e., that fraction that has the longest residence time in the atmosphere and that has the largest impact on visibility degradation). The $PM_{2.5}$ fraction often behaves differently than the coarse fraction with respect to dispersion and deposition.

Recommendations based on Finding #8

(8.1) Conduct field tests to quantify the vertical $PM_{2.5}$ flux for fugitive windblown dust and mechanically suspended road dust.

Finding #9: Air quality models need to integrate meteorology and the emissions processes Regional scale air quality models need to integrate all the appropriate meteorological processes with the fugitive dust emissions generation processes. Meteorological parameters used for estimating transport and dispersion should be combined with the fugitive dust emission estimates for both mechanically-generated and wind erosion-generated conditions. For example, to model windblown dust, if time-averaged wind speeds are used, it is important to account for smaller time-scale wind gusts. The use of air quality dispersion models that account for injection heights and deposition losses should allow one to distinguish the relative impact from near field (i.e., local) emissions versus far field (i.e., regional) emissions on ambient air quality.

Recommendations based on Finding #9
(9.1) Reconcile model predictions with measurements by incorporating an interim method for accounting for near source removal of particles into regional models. This will involve running air quality dispersion models utilizing estimates of the vertical fugitive dust emissions flux rather than the local-scale horizontal flux as inputs for the model for those cases where the model(s) predicted ambient concentrations that were considerably larger than the observed downwind concentrations.
(9.2) Incorporate more refined particle removal estimation methods into regional models as they become available.

Finding #10: Disturbed surfaces produce significantly more fugitive dust than undisturbed surfaces In general, undisturbed surfaces produce much less dust than disturbed surfaces because the undisturbed surface usually requires considerably higher wind speeds to become a significant emission source. A surface having a lower threshold velocity produces much more dust than a surface having a higher threshold velocity. The primary influence of disturbing a surface is to lower the threshold velocity and increase dust emissions.

Recommendations based on Finding #10
(10.1) Catalog existing studies and conduct studies to determine if different surfaces are supply-limited or supply-unlimited (i.e., unlimited particle reservoir).
(10.2) Conduct studies to determine the effect that different kinds of disturbance have on the aerodynamic roughness height of different surfaces.

Finding #11: Receptor models may be used to distinguish contributions from different sources of fugitive dust Uncertainties in fugitive dust emission estimates for crustal materials can be estimated and reduced to acceptable levels by reconciliation with ambient measurements. Variations in particle size and chemical composition of the fugitive dust at a receptor site as well as the temporal and spatial variations for multiple receptor sites can be used to indicate the spatial scale of the sources, the portion of the day when fugitive dust contributions are large, and whether the fugitive dust is wind generated. Additional chemical and physical measurements of the ambient aerosol at receptor sites may shed some light on specific dust sources. Receptor measurements should be used with model estimates to evaluate modeled source contributions and to focus inventory improvement efforts.

Recommendations based on Finding #11

(11.1) Examine the IMPROVE data base for enrichments of geological elements relative to median ratios or to soil compositions typical of the areas in which samplers are located; and determine the extent to which different elemental ratios are observed, outside of natural variability, and determine the extent to which these are correlated with elevated $PM_{2.5}$ soil or coarse particle mass concentrations.

(11.2) Examine DRUM impactor measurements for soil related species from IMPROVE special studies; identify typical size distributions and deviations from those distributions; and relate outliers to $PM_{2.5}$ and coarse mass concentrations.

(11.3) Examine nephelometer light scattering data from the IMPROVE network; and determine the extent to which spikes are observed and whether these can be related to fugitive dust or other sources.

(11.4) Critically review previous studies of microscopic, isotopic, organic, and other measurement methods with respect to their ability to distinguish among different land forms and fugitive dust emitters; and determine which methods can be practically applied to source characterization and ambient sampling.

(11.5) Design and implement a systematic source profile measurement program for geological samples obtained from open surfaces upwind of Class I areas in the West; examine existing soil and geological maps to identify areas that have suspension potential; obtain sufficient dust quantities from these areas for a variety of analytical methods; suspend and sample these quantities through PM_{10} and $PM_{2.5}$ size-selective inlets; analyze by the selected methods; and apply appropriate mathematical tests to determine the degree to which profiles are collinear or are dissimilar enough to distinguish between fugitive dust source types and areas.

(11.6) Implement high time resolution methods, preferably ones that are specific to coarse particles, at several IMPROVE sites and between these sites and suspected source areas; examine these to determine when dust events occur, their durations, and their contributions to 24-hour averages; and use this information to focus emissions inventory efforts.

(11.7) Characterize the chemical composition of the coarse aerosol fraction collected at selected IMPROVE sites and use the chemical mass balance (CMB) receptor model to try to differentiate between different fugitive dust source categories.

(11.8) Identify and develop methodology for resolving local sources of fugitive dust from distant sources.

(11.9) Identify and develop methodology for resolving road dust from windblown dust.

Time Frame for Implementing Recommendations

The recently enacted Regional Haze rule requires that many states complete their SIPs by December 2003. In order for these states to meet the deadline, they will need to implement many of the expert panel's recommendations over the next two years. These recommendations have been grouped into three categories: (1) short term recommendations that can realistically be implemented in the next 12 months, (2) intermediate term recommendations that should be implemented in the next 12 to 18 months, and (3) long-term recommendations that should be implemented in the next 18 to 30 months.

Schedule for Implementing Expert Panel Recommendations

Recommendation	Short Term (next 12 months)	Intermediate Term (next 18 months)	Long Term (18 to 30 months)
1.1		X	
1.2		X	
1.3		X	
2.1		X	
2.2			X
3.1		X	
3.2		X	
3.3		X	
3.4		X	
4.1	X		
4.2			X
5.1			X
5.2		X	
6.1		X	
6.2		X	
6.3			X
7.1		X	
7.2		X	
7.3		X	
7.4			X
8.1		X	
9.1		X	
9.2			X
10.1		X	
10.2		X	
11.1	X		
11.2		X	
11.3		X	
11.4		X	
11.5		X	
11.6			X
11.7	X		
11.8		X	
11.9		X	

1. INTRODUCTION

1.1 Background

The Clean Air Act Amendments of 1990 required EPA to establish the Grand Canyon Visibility Transport Commission (GCVTC or Commission) for the region including Grand Canyon National Park. The Commission was charged with reviewing scientific, technical and other information and with submitting a report to EPA by 1995 on measures that could be implemented to improve visibility. Due to the regional nature of the problem, EPA expanded the scope of the Commission's focus to include not only Grand Canyon National Park, but also 15 other Class I areas on the Colorado Plateau. Among other things, the Commission developed an emission inventory and applied an air quality model to estimate the contribution of those emissions to visibility impairment. It also recommended several potential emission reduction measures and identified areas for future investigation. The Commission's report (GCVTC 1996) that summarizes these activities serves as a basis for the current understanding of the visibility in and near the Colorado Plateau.

In 1997, states and tribes formed a stakeholder-based successor organization to the GCVTC known as the Western Regional Air Partnership (WRAP). The WRAP's purpose is to continue the work of the GCVTC in developing and planning programs that can contribute to reducing emissions and improving visibility in the west. The administrative responsibilities of WRAP lie within the Western Governors' Association (WGA)and National Tribal Environmental Council. Following the promulgation of the Regional Haze Rule, EPA designated the WRAP as the organization that would be funded to coordinate the regional planning effort among the western states. While the WRAP can help facilitate the implementation of regional haze regulations and recommend regional approaches to improving air quality and reducing regional haze, the responsibility for implementing any or all of the recommendations of the WRAP lies with individual states and tribes.

The Research and Development (R&D) Forum of the WRAP has the responsibility to advise the WRAP on technical issues and, if necessary, recommend avenues of research to address these issues.

1.2 Statement of the Problem

The Executive Summary of the GCVTC Report (1996) includes a section on road dust which reads:

> "*The Commission's technical assessment indicates that dust is a large contributor to visibility impairment on the Colorado Plateau. As such, it requires urgent attention. However, due to considerable skepticism regarding the modeled contribution of road dust to visibility impairment, the Commission recommends further study in order to resolve the uncertainties regarding both near-field and distant effects of road dust, prior to making remedial action. Since this emission source is potentially such a significant contributor, the Commission feels that it deserves high priority attention and, if warranted, additional emissions management actions."*

The relative importance of this source category is evidenced by visibility and particle monitoring data collected in western sites since 1988 (Sisler et. al., 1993, 1996, 2000). At several of these sites, particularly on the Colorado Plateau and in the Desert Southwest, suspended coarse particulates (2.5 μm to 10 μm in diameter) and fine soil particles (2.5 μm and smaller) are major contributors to visibility impairment. For example, the annual average apportioned fine mass and undifferentiated coarse mass for the Colorado Plateau sites, shown in Figure 1-1, account for nearly 70% of the particle mass (10 μm and smaller). Figure 1-2 shows the relative contribution to visibility impairment due to the constituents shown in Figure 1-1 and Rayleigh clear sky scattering. Note that the coarse material and fine soil account for about 20% of the average visibility impairment.

To assess this source category the Commission used the best available emission factors to develop an emission inventory for dust (fine soil and coarse material). Air quality model applications used this inventory to estimate ambient concentrations and the dust contribution to visibility impairment. However, the calculated ambient concentrations and the associated extinction due to dust were found to dominate the concentration estimates from all sources combined and to compare poorly with ambient measurements. Therefore, the modeled ambient concentrations were "scaled" to provide more reasonable estimates. Still, considerable uncertainty remained on the relative contributions of dust derived from nearby and distant area sources. Therefore, the Commission felt there was sufficient cause to discount many of the assessment results, including contributions to visibility impairment that were attributed to windblown and resuspended road dust. Specifically, the Commission identified the near-field and far-field effects of dust on visibility from paved and unpaved roads in Class I areas as an area requiring considerable further study.

1.3 Objectives

Because the Commission believed that there was a problem with the emission factors used for developing the emission inventory for dust, the WRAP's R & D Forum initiated an effort to better understand the role of dust on visibility impairment for the Colorado Plateau.

The specific objectives of this effort are to:

- Summarize the findings and recommendations of a panel of air quality experts regarding the best methodology for estimating emissions of fugitive windblown and mechanically resuspended road dust applicable for regional scale air quality modeling, and
- Make recommendations for future research activities to generate improved fugitive dust emissions estimation techniques applicable for regional scale air quality modeling.

1.4 Technical Approach

To meet the specific objectives identified above, the Western Governors' Association (WGA) issued a contract to Countess Environmental (CE) to accomplish the following three tasks:

- Conduct a literature search for relevant reports and publications,
- Convene an expert panel to assess the subject matter,
- Host a public workshop to present the experts' findings and recommendations.

Figure 1-1. Annual Average Mass Concentration Budget for the Colorado Plateau (1996-1999)

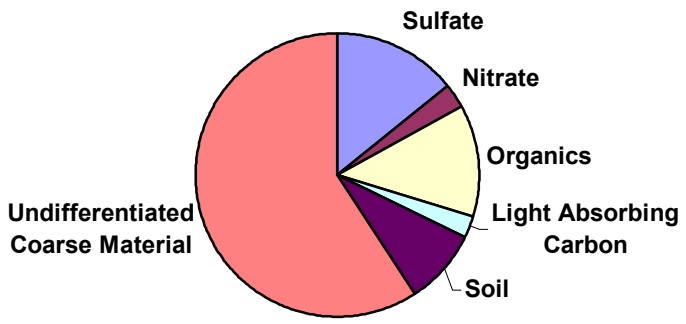
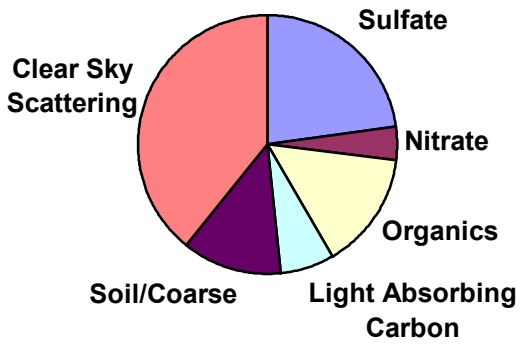


Figure 1-2. Annual Average Aerosol Extinction Budget for the Colorado Plateau (1996-1999)



The approach adopted for each task is outlined below.

1.4.1 Literature Search

This first task involved compiling a master list of titles of existing literature on fugitive windblown and mechanically resuspended road dust emissions and their contribution to regional air quality in the Western United States. Richard Countess, as Principal Investigator, recruited six nationally recognized experts (see Section 1.4.2) with expertise in different disciplines to identify existing data. These experts were asked to critically assess the relevance, validity, and uncertainties of published and unpublished data.

The lists of titles of relevant publications and reports compiled by the experts were augmented with relevant reports from CE's files, Desert Research Institute's report titled "Reconciling Urban Fugitive Dust Emissions Inventory and Ambient Source Contribution Estimates: Summary of Current Knowledge and Needed Research" (Watson et. al., 2000), and documents prepared by or on behalf of the Grand Canyon Visibility Transport Commission that addressed fugitive dust and its impact on visibility. Each expert (as well as the WGA project officer) provided CE with copies of reports that are not readily available in the technical literature (i.e., reports that appear in the commercial and government report literature). These reports were assembled and made available to the experts for their review.

1.4.2 Expert Panel

For this task, CE, in consultation with members of the WRAP R&D Forum, recruited a panel of six experts to identify the best methods for estimating fugitive windblown and mechanically resuspended road dust emissions appropriate for regional scale air quality modeling. As part of their assessment, these experts identified information gaps and uncertainties, resolved issues to the extent possible, and recommended future research activities to address these gaps and uncertainties.

The expert panel consisted of the following individuals from both the private and public sectors:

Bill Barnard, Harding ESE, Inc.
Candis Claiborn, Washington State University
Dale Gillette, National Oceanographic and Atmospheric Administration
Doug Latimer, USEPA Region VIII
Tom Pace, USEPA
John Watson, Desert Research Institute

with technical support from the co-chairs of the WRAP R&D Forum, Mark Scruggs (National Parks Service) and C.V. Mathai (Pinnacle West Capital Corp.).

To expedite the assessment process, the experts were provided with a list of hypotheses and issues that needed to be resolved to meet the WRAP's objectives based on a list compiled by CE, members of the R&D Forum, and the six experts. The panel was asked to evaluate each hypothesis/issue, provide support for or against each, resolve any issues that arose to the extent possible, comment on the strengths and weaknesses of the methods used for estimating fugitive dust emissions appropriate for regional scale modeling, identify areas where information is lacking, and suggest ways to fill these information gaps.

Assessment by the experts was performed in consultation with Dr. Countess and the R&D Forum co-chairs. An expert panel meeting, held in Ventura, CA, on September 11th and 12th, 2000, gave each expert the opportunity to present his/her position on various topics and respond to questions from his/her peers. In preparation for this meeting, each panel member submitted to CE a document annotated with references supporting their conclusions regarding each hypotheses or issue. These documents were then distributed to each expert prior to the panel meeting. The agenda for the panel meeting included allocating a specific block of time to discuss each hypothesis or issue. The expert selected to take the lead in discussing a specific topic presented his/her position on that specific topic, and then the panel was given the opportunity to query that speaker. At the conclusion of discussion on all the topics, the experts developed a list of eleven findings, summarized what information gaps still existed and produced a list of recommendations for further investigation. This report compiles and integrates the contributions from all the experts. The findings are listed here and discussed in detail in Sections 2 through 12 of this report.

Finding #1: Not all suspendable particles are transported long distances
Finding #2: Regional scale vertical flux is smaller than the local-scale fugitive dust flux
Finding #3: Fugitive dust emission factors need to be appropriate
Finding #4: Source activity levels need to be accurate
Finding #5: Fugitive dust emissions are not continuous processes
Finding #6: Annual fugitive dust emission inventories are not sufficient
Finding #7: Spatial allocation of fugitive dust emissions is important
Finding #8: The fine fraction of fugitive dust emissions is not adequately characterized
Finding #9: Air quality models need to integrate meteorology and the emissions processes
Finding #10: Disturbed surfaces produce significantly more fugitive dust than undisturbed surfaces
Finding #11: Receptor models may be used to distinguish contributions from different sources of fugitive dust

1.4.3 Workshop

A public workshop was held on December 14, 2000, in Las Vegas, NV, at which the experts discussed their findings and recommendations for developing a scientifically robust emission inventory for fugitive windblown dust and mechanically resuspended road dust that contribute to the degradation of regional air quality and visibility in Class I areas of the West. The experts also summarized the limitations of the recommended methodology and identified information gaps.

Topics that were presented at the workshop included:

- Emission factors and source activity levels
- Reservoir of suspendable particles
- Spatial and temporal variation in emissions
- Intermittent emissions
- Vertical flux versus horizontal flux
- Local scale flux versus regional flux
- Disturbed versus undisturbed surfaces

- Regional transport
- Particle loss mechanisms
- Air quality and source/receptor models

1.5 Structure of this Report

The eleven findings developed by the expert panel are presented in Sections 2 through 12 of this report. In each section, the finding is first summarized. Then, the technical support for the finding is presented, followed by a discussion of the limitations and information gaps, and, finally, recommendations for future work are presented. Section 4 is broken down into two subsections. The first addresses emission factors for windblown dust, and the second addresses emission factors for mechanically resuspended paved and unpaved road dust. A bibliography of all the relevant reports and publications identified during the course of this project appears at the end of the report.

2. NOT ALL SUSPENDABLE PARTICLES ARE TRANSPORTED LONG DISTANCES (FINDING #1)

Only a fraction of suspendable particles are regionally transportable particles. Ground level emissions of mechanically generated particles are likely to be removed near the source, due to gravitational settling as well as impaction on nearby obstacles, with large particles having a greater removal rate than small particles. The initial vertical energy associated with mechanically generated particles is typically short-lived and unsustained. In the absence of violent winds with large vertical components (such as those in dust devils or thunderstorms) or significant solar heating of the ground to cause upward diffusion due to large turbulent eddies, there is little, if any, residual or continuing source of energy to sustain vertical motion and transport of these emissions away from the source. For winds accompanied by gusty conditions or high turbulence, windblown dust emissions may be lofted vertically to great heights above the ground by the sustained energy provided by the vertical component of the wind and transported long distances from the source.

2.1 Technical Support of Finding #1

Figure 2-1 shows the cumulative horizontal PM_{10} flux at different elevations above the ground for unpaved roads (Watson et. al., 2000). Similar results for paved roads, construction sites and bare soil (Watson et. al., 2000) indicate that approximately half of the mass does not attain elevations exceeding 1 to 2 meters above ground level (agl) near the source. 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.

Figure 2-2 shows the estimated depletion with time of 2.5 µm and 10 µm particles that are uniformly mixed through layers of 1, 10, and 100 meter thickness. Figure 2-2 actually shows upper limits on residence times because it assumes particles to be uniformly mixed throughout the layer at all times and only gravitational settling is considered. Gravitational settling usually causes concentrations near ground level to exceed those at the top of the layer, and turbulent eddies often include negative vertical velocity components that add to gravitational settling. Even with these conservative assumptions, Figure 2-2 shows that half of the 10 µm particles mixed within the first meter are removed after ~3.5 minutes, and that half of the 2.5 µm particles in this layer are gone after an hour. This assumes that these near-surface particles do not encounter low-level obstructions in their pathways. Less than 10% of the 10 µm particles remain after 12 minutes, with 90% of the 2.5 µm particles depleted after 3.5 hours. Fugitive dust lifetimes would be short compared to a regional effect even if the suspended particles were evenly mixed through a horizontal plane of 10-meter depth perpendicular to the wind direction. Figure 2-2 shows that there is a substantial difference in depletion with respect to particle size; the ratio of $PM_{2.5}$ to coarse (PM_{10} minus $PM_{2.5}$) concentrations would increase substantially within a few hours after suspension, and this ratio may be a good indicator of local versus distant contributions. A 1 m/s wind speed results in a transport distance of 3.6 km in one hour. In an average 5 m/s wind, only 10% of the 10 µm particles uniformly mixed throughout a 10 meter thick layer would travel more than 36 km from the source, while 10% of the 2.5 µm particles could achieve distances of nearly 600 km. As noted above, given the assumptions of Figure 2-2

Figure 2-1. Cumulative horizontal PM_{10} flux at different downwind elevations above different unpaved roads (data from Cowherd, 1999; plot from Watson et. al., 2000).

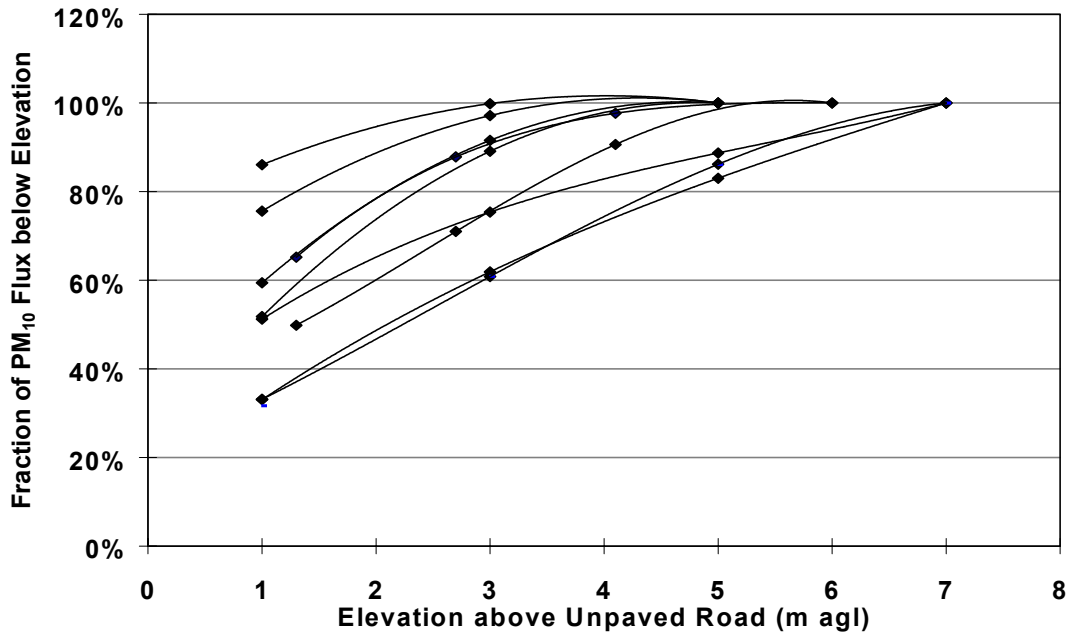
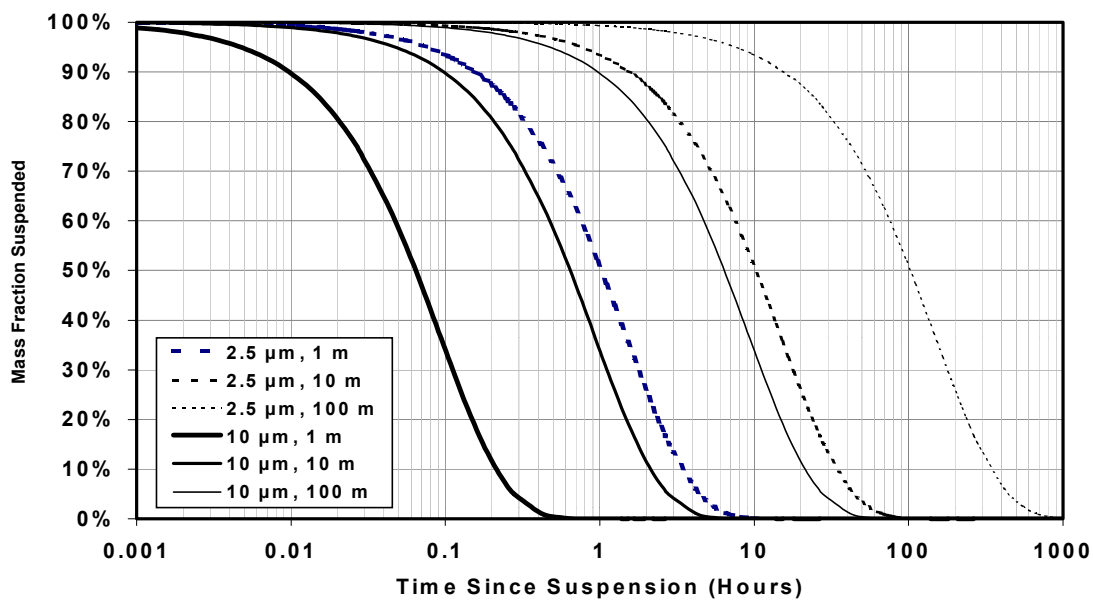


Figure 2-2. Attenuation of PM_{10} and $PM_{2.5}$ mass concentrations with time and vertical mixing volume (adapted from Watson and Chow, 2000). Assumes stirred tank model of Hinds (1999) in which particles are homogeneously redistributed throughout the mixed layer at each time step.



these estimates are positively biased for most suspension and transport situations. They indicate that $PM_{2.5}$ emissions are more relevant to regional inventories than are PM_{10} emissions from fugitive dust sources, but empirical emissions factors for this size fraction are lacking for fugitive dust source categories.

The rapid attenuation of PM_{10} concentrations due to deposition and dispersion downwind of an unpaved road is illustrated in Figure 2.3. While there is substantial scatter in the ratios of downwind to roadside concentrations, it is clear that PM_{10} is attenuated by ~90% within a very short distance of only 50 meters from an unpaved roadside.

Watson et. al., (2000) summarize published control measure effectiveness evaluations for urban fugitive dust sources such as paved and unpaved roads, bare ground, and construction sites. Control measures typically attempt to reduce the reservoir of available dust by removal, enclosure or stabilization, or reduction of the activities that create the reservoirs and suspend the dust into the atmosphere. Quantitative tests of emissions reduction efficiencies are relatively few and yield widely varying results, ranging from 0% to >90% for similar control measures. Much of this variability is probably due to the staying power of the controls (e.g. unpaved road dust suppressants) and the diligence with which they are implemented (e.g. watering of continuously disturbed surfaces at construction sites). Control efficiencies are specific to PM_{10} or TSP (total suspended particulate, nominally particulate material with an aerodynamic diameter of 30 μm or smaller), and their effectiveness for the $PM_{2.5}$ fraction that is most likely to transport from cities to Class I areas has not been evaluated. National Parks contain, and are often surrounded by, paved and unpaved roads that are not usually subject to control measures. These roads may be surrounded by obstructions, such as forests, rocks, and canyon walls that inhibit the long-range transport of material.

Obstructions on the upwind or downwind side of an exposed surface have two effects on fugitive dust emission and transport. Upwind obstructions lower wind speeds and turbulence near the ground. This decreases the probability of achieving a suspension threshold velocity at the surface, minimizes the upward lift of particles that are suspended mechanically or by wind, and limits the downwind transport distance of suspended particles because the wind speed is lower. The effectiveness of these obstructions depends on their height, length, permeability (i.e., the fraction of their area which is open, typically the density of leaf and branch cover), and surface roughness characteristics of the surrounding area. Figure 2-4 shows one of the many studies summarized by van Eimern et. al. (1964) that examine how wind speeds and turbulence change behind shelterbelts and wind fences that are often used to reduce crop erosion, reduce snow blow, and minimize deposition on roads during dust storms. Permeability is an important parameter; Figure 2-4 indicates that an obstruction with a medium permeability has the longest fetch with a greater than 60% reduction in wind speed Although a solid wall (i.e., a very dense obstruction) provides a precipitous reduction in wind speed, the effect does not persist for long distances downwind of the obstruction. The implication of Figure 2-4 is that a thin row of trees of medium permeability upwind of a dust source provides greater downwind reductions of wind speed than a dense row. For deciduous trees, permeability changes substantially with season as they are leafless during winter. Snow accumulations on conifers increases their permeability, but fugitive dust is not usually an issue when surfaces are snow covered. The exception is for roadways to which wintertime de-icing materials are applied.

FIGURE 2-3. Attenuation of PM_{10} concentrations with distance from an unpaved road (Watson et. al., 1996)

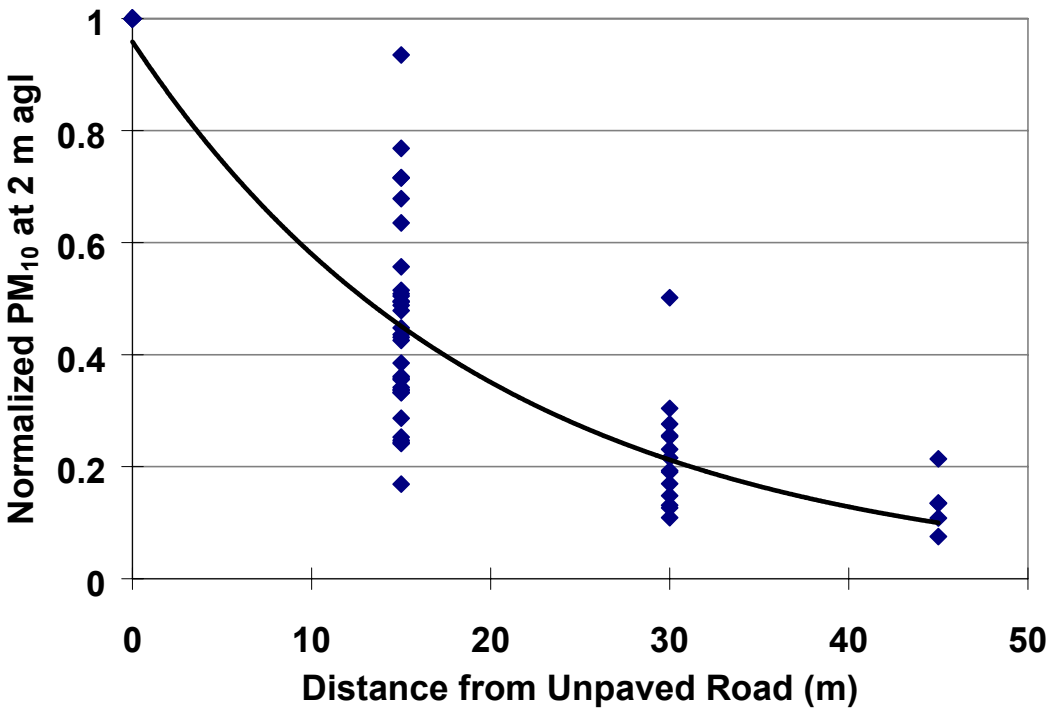
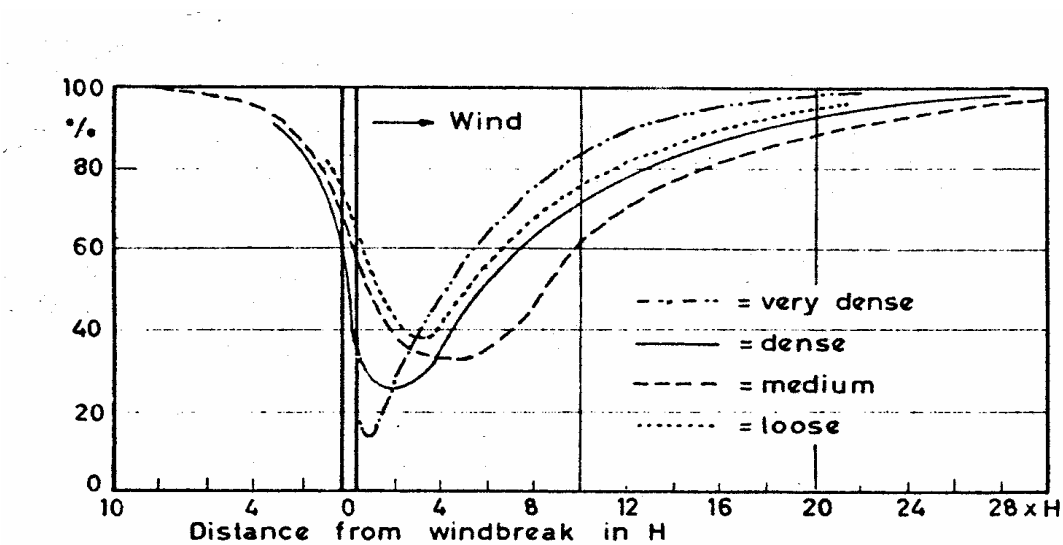
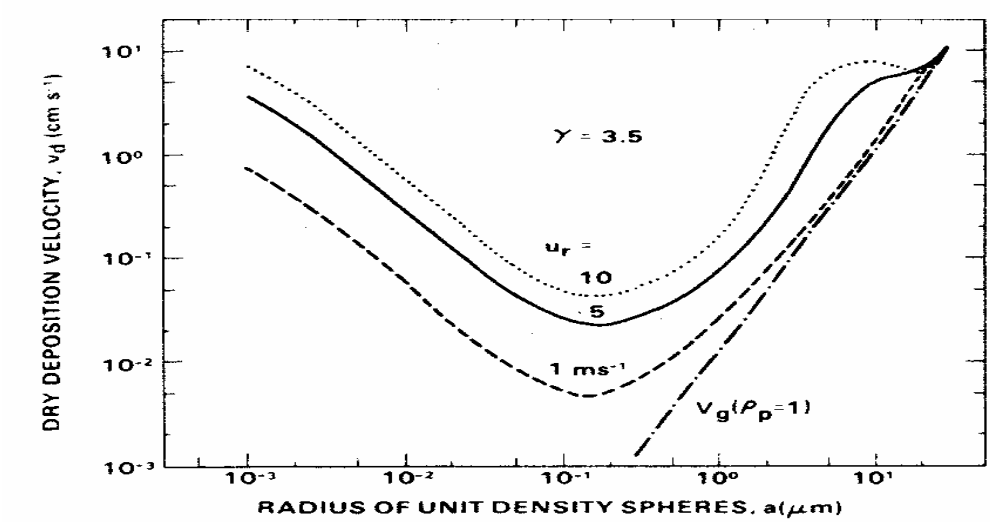


Figure 2-4. Attenuation in wind speed downwind of an obstruction (graph by Naegeli, 1946 as reported by van Eimern et. al., 1964). The vertical axis represents the reduction in wind speed with respect to the speed upwind of the obstruction. The horizontal axis shows distance in units of height (H) of the obstruction. Different curves represent different permeabilities of the obstruction.



Downwind obstructions will attenuate wind speeds, and therefore reduce transport distances; in addition they also provide more surfaces for deposition of suspended particles (Gupta and Kapoor, 1992; Kapoor and Gupta, 1984). A deposition theory with reasonable assumptions developed by Slinn (1982) is compared with gravitational deposition in Figure 2-5 for various parameters (height, permeability, surface roughness, friction velocity) typical of a Eucalyptus forest. This model could be adapted via specification of these variables for a variety of vegetative and non-vegetative obstructions. Note first that deposition on a downwind obstruction always exceeds gravitational deposition. This confirms that the residence times illustrated in Figure 2-2 are upper limits, and that lifetimes are probably much lower than those indicated in the figure. For a 5 m/s wind speed, the deposition rates of 2.5 μm and 10 μm diameter particles are more than five times the gravitational settling rate. This indicates that the increase in transport distance owing to higher wind speeds is nearly compensated by increased deposition within the obstruction barrier. Figure 2-5 also shows that deposition reaches a minimum for particles with 0.3 to 0.5 μm diameters, thereby allowing a larger fraction of such particles to bypass the barrier. This size range is dominated by secondary aerosols and combustion aerosols such as those in vehicle exhaust that might accompany suspended road dust. Downwind obstructions would preferentially pass these fine particles while removing the larger geological particles. This might be part of the explanation for finding higher ratios of combustion to geological particles in ambient samples relative to those in emissions inventories.

Figure 2-5. Estimated changes in deposition velocity (v_d) as a function of particle radius for different wind speeds (u_r) through a Eucalyptus forest (Figure 5 from Slinn, 1982). The factor γ includes parameters related to the obstruction height, permeability, and turbulence that can be determined for representative surface covers; v_g is the gravitational settling velocity.



2.2 Limitations

Robust theoretical and empirical models that describe how wind speeds and turbulence change downwind of obstructions are lacking. Most of the available information is empirical and focuses on wind breaks used in agricultural operations. Information on natural obstructions such as desert scrub brush, undulating terrain, and rocky outcroppings is lacking.

2.3 Information Gaps

Major information gaps include:

- $PM_{2.5}$ flux measurements at different elevations above dust reservoirs. Flux measurements such as those shown in Figure 2-1 have been developed for PM_{10} and TSP, but it is likely that the results will be different for smaller particles.
- Classifications of upwind and downwind obstructions with respect to height, permeability, surface roughness, and other variables that affect lee side wind speed and particle deposition.

2.4 Recommendations

Recommendations for future research studies include:

- $PM_{2.5}$ upwind/downwind experiments. Experiments similar to those used to develop TSP and PM_{10} horizontal flux measurements should be conducted around roadways and exposed surfaces to determine the heights at which different $PM_{2.5}$ quantities pass. Owing to lower deposition velocities and lower suspension energies, it is likely that the $PM_{2.5}$ vertical flux distribution is much more uniform than the PM_{10} or TSP vertical flux distributions.
- Assignment of relevant parameters (e.g., barrier height, length, permeability, surface roughness and friction velocity) to ground cover classifications that can be determined from GIS data bases, such as the USGS groundcover data base. These parameters are needed for different seasons to account for changes such as leaf growth and depletion. These parameters can then be used in emissions models that associate them with groundcover data to estimate how wind speeds are attenuated and to estimate more accurate deposition velocities that remove particles from long-range transport.
- Expand the theoretical and empirical models developed for trees to other obstructions, such as desert scrub and buildings; and implement these in emissions models to better simulate wind speed, turbulence, and removal due to obstructions surrounding dust reservoirs.
- Develop preliminary estimates of deposition velocity for relevant parameters (e.g., barrier height, length, permeability, and surface roughness) and link to a gridded database of land cover across the Western US.

3. REGIONAL SCALE VERTICAL FLUX IS SMALLER THAN THE LOCAL-SCALE FUGITIVE DUST FLUX (FINDING #2)

The regional scale vertical fugitive dust flux for a large scale transport model applies only to particles that are not deposited in the same grid cell from which they are emitted. Since a portion of the particles are deposited within the same grid cell from which they were emitted, the effective regional scale vertical flux of fugitive dust particles is smaller than the local scale vertical flux. The ratio of the effective regional scale vertical flux to the local scale vertical flux is a function of the friction velocity of the surface and the deposition velocity of the different size particles.

3.1 Technical Support of Finding #2

A "box" model for relating regional scale vertical dust flux to local scale fugitive dust emissions, where the vertical fugitive dust flux constitutes a fraction of the entire surface of the grid cell (as opposed to a uniform vertical flux that covers the entire surface of the grid cell), is presented below. Using the model it is possible to scale particle emissions from small areas (such as1 km by 1 km grid cells) to the larger grid sizes used for regional inventories. Application of this box model in the preparation of such inventories accounts for particles that are suspended and deposited within the emission grid cell, and therefore should not be included in the regional transport calculation.

General formulation of the model A case using a "lumped control volume" approach that considers fugitive dust emissions generated from a road surface (Note: this model is also applicable to any dust suspension surface or activity) is presented in Figure 3-1. A dirt road exists at the right side of the control volume; to the left of the road is a surface that is grass or shrub-covered and does not emit particles. The top of the control volume is the surface through which the vertical flux of particles pass. By letting the road be directed into and out of the page while wind is directed from right to left, two-dimensionality or symmetry in the direction into the page (i.e., fluxes are equal into and out of the page) is applicable.

A conservation of mass equation can be written for the control volume shown in Figure 3-1 as:

$$dM/dt + dm_{up}/dt + dm_{depos}/dt + dm_{ambout}/dt - dm_{ambin}/dt - dm_{road}/dt + dm_{ceil}/dt = 0 \qquad (3\text{-}1)$$

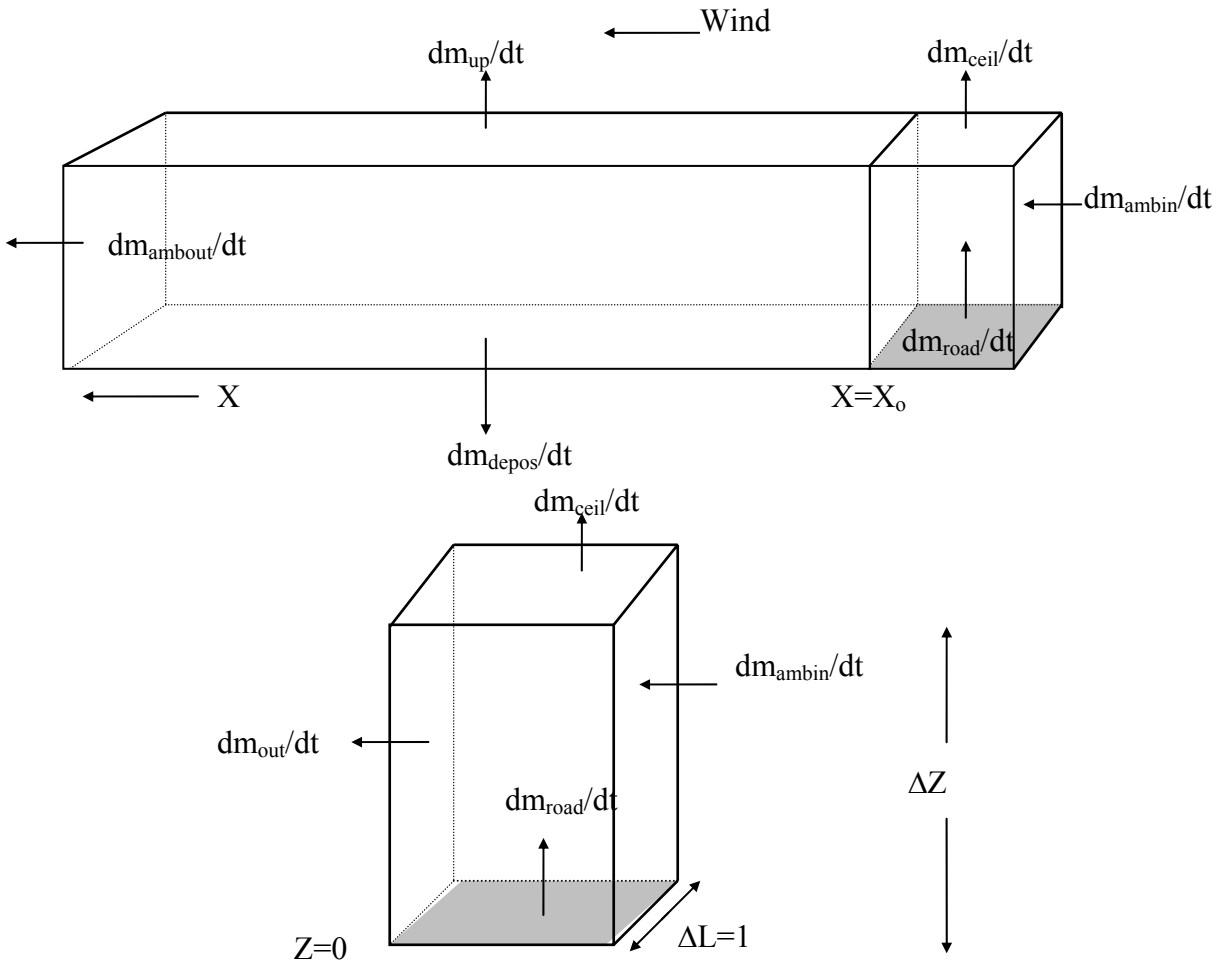
where, M is the mass of particles in the control volume (CV),

dm_{up}/dt is the mass per unit time passing through the top of the CV,
dm_{depos}/dt is the mass per unit time depositing to the floor of the CV,
dm_{ambout}/dt is the mass per unit time passing out of the CV through the left wall,
dm_{ambin}/dt is the mass per unit time passing into the CV from the right wall,
dm_{road}/dt is the mass per unit time emitted from the road, and
dm_{ceil}/dt is the mass per unit time passing out of the ceiling directly above the road.

A simplifying assumption is that of steady state emissions with dM/dt = 0. For this case:

$$dm_{up}/dt = - dm_{depos}/dt - dm_{ambout}/dt + dm_{ambin}/dt + dm_{road}/dt - dm_{ceil}/dt \qquad (3\text{-}2)$$

Figure 3-1. Control Volume for Fugitive Dust Model Depicting Vertical and Horizontal Flux



Relationship of mass per unit time to the horizontal mass flux from the road A subvolume of the control volume is shown in Figure 3-1 as that part that contains the road but no area downwind of the road. The left wall of this part of the control volume is the surface through which all of the dust emitted from the road passes. None of the road dust is part of the ceiling flux (dm_{ceil}/dt) Before deposition occurs, the horizontal flux through the wall immediately downwind of the road is equal to the vertical flux from the road. For the condition of steady state, a description of the conservation of mass for this part of the control volume is:

$$dm_{road}/dt = dm_{out}/dt - dm_{ambin}/dt + dm_{ceil}/dt \tag{3-3}$$

where dm_{out}/dt is the horizontal mass per unit time passing out through an imaginary wall just left of the left edge of the road reaching from the surface to the top of the control volume, and dm_{ceil}/dt is the mass per unit time passing through the ceiling of the control volume. Because the road dust is usually the overwhelming source of dust (i.e., $dm_{road}/dt >> dm_{ambin}/dt + dm_{ceil}/dt$), the horizontal flux ($dm_{out}/dt$) for these conditions is approximately:

$$dm_{road}/dt = dm_{out}/dt \tag{3-4}$$

which is the horizontal mass flux of dust from the road.

Deposition on the floor and vertical flux of dust through the ceiling of the control volume downwind of the road The total loss of material diffusing vertically through the ceiling and depositing on the floor of the control volume to the left of the road is estimated by first calculating the effective concentration at position x to the left of x_0 (i.e., the left edge of the road) and then utilizing the following equation:

$$V\frac{dC(x)}{dx} = \frac{-C(x)[V_d + K]}{\Delta z} \tag{3-5}$$

where V is the wind speed that carries the dust through the control volume (CV); C(x) is the concentration of dust mass at position x in the CV; x is the horizontal position in the CV (increasing to the left); z is the vertical position in the CV (increasing from the floor to the ceiling); V_d is the deposition velocity; and K is a coefficient having the dimensions of velocity.

A solution to Equation (3-5) is:

$$C(x) = \frac{\frac{dm_{road}}{dt}}{V\Delta z\Delta L}\exp-\left[\frac{[V_d + K]}{V\Delta z}(x - x_o)\right] \tag{3-6}$$

Multiplying C(x) by $V_d\ \Delta L$ (i.e., the deposition velocity times a unit length roadway) and integrating with respect to x from $x = x_0$ to $x = \infty$ gives:

$$\frac{dm_{depos}}{dt} = \frac{V_d\frac{dm_{road}}{dt}}{(V_d + K)} \tag{3-7}$$

By making the approximation that $dm_{ambout}/dt - dm_{ambin}/dt \approx 0$ and that dm_{ceil}/dt is negligible, Equation (3-2) reduces to:

$$\frac{dm_{up}}{dt} = \frac{dm_{road}}{dt} - \frac{dm_{depos}}{dt} \tag{3-8}$$

Watson et. al. (1996) observed that dm_{ambout}/dt - dm_{ambin}/dt approaches zero at a distance x of about 100 meters downwind of an unpaved roadway in the absence of intervening dust sources.

Ratio of vertical flux of road dust into the atmosphere to the horizontal flux of road dust This ratio, expressed by the symbol Φ, is given by:

$$\Phi = \frac{\frac{dm_{up}}{dt}}{\frac{dm_{road}}{dt}} = \left[1 - \frac{V_d}{(V_d + K)}\right] = \frac{K}{(V_d + K)} \tag{3-9}$$

Φ can be estimated by making an approximation based on the data presented by Gillette (1974) that $K = 0.08\ u_*$, where u_* is the friction velocity. Using Equation 3-9 and this expression for K, one can derive values of Φ for typical values of V_d and u_*. Values of Φ are shown in Figure 3-2 using values for V_d versus size and environmental conditions given by Slinn (1982).

Conclusion Equation 3-9 explains a large part of why observed fugitive dust concentrations are smaller than those estimated by regional scale models that use the entire amount of dust emitted by roads or other surfaces without adjusting for downwind particle loss within the emissions grid cell. Because dust is removed by dispersion and deposition to surfaces close to the source, uncorrected dust vertical fluxes lead to overestimating dust concentrations downwind of the source. Other effects that are expected based on the results shown in Figure 3-2 are:

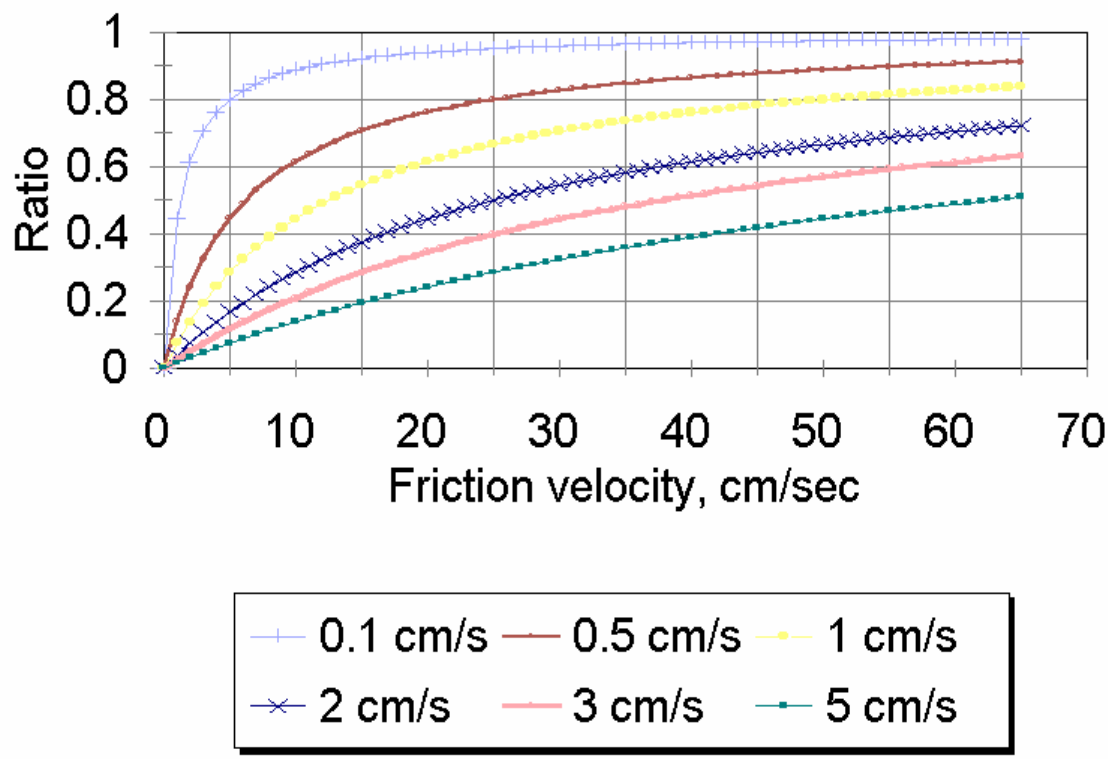
- For low wind speeds, little dust is generated above the surface level of the roadway. This implies that the thickness of the first layer of a regional air quality model is important to the transport of dust. This agrees with observations by Johnson et. al. (1992).
- For industrial sources of fugitive dust associated with loading/unloading soil or aggregate (such as unloading sediment with a front-end loader), Figure 3-2 overestimates Φ since horizontal diffusion would make more dust available for deposition to the ground. A correction for this effect would involve reducing Φ.
- For wind erosion, the threshold friction velocities are typically larger than 19 cm/s. Therefore, for windblown dust the regional scale vertical flux is expected to be a large fraction of the local scale flux.
- Dust devils (a special case of wind erosion) occur when the overall friction velocity is fairly low, but the height of the initial dust being generated is very high. For these conditions, the vertical flux dm_{up}/dt is virtually the same as that of the dust being generated. Consequently, dust devils should be considered to be effective sources of dust. Gillette and Sinclair (1989) give an excellent discussion of dust devil dust fluxes.

3.2 Limitations

Limitations depend on the extent to which assumptions for this simplified box model are attained in actual practice. These limitations include:

- The flux through the ceiling above the road, dm_{ceil}/dt, may not be negligible.
- The flux upwind of the road, dm_{ambin}/dt, may not be negligible compared to the flux from the road. If the flux upwind from a source is not negligible, a regional model must accommodate carrying the dust from upwind to downwind.
- The approximation of a well mixed dust concentration from the ground to the top of the control volume may not be a good description of the natural situation.

Figure 3-2. Ratio of Regional Scale Vertical Flux to Local Scale Fugitive Dust Flux as a Function of Friction Velocity for Various Values of Deposition Velocity.



- The assignment of the deposition as V_d C(x) may be different from the deposition taking place in the natural situation.
- Non-steady state conditions may not be observed in the natural situation.
- The downwind flux, dm_{ambout}/dt, within a few hundred meters to a kilometer of the source may not be approximately equal to the upwind flux, dm_{ambin}/dt. In this case the model would not be applicable to regional scale applications. Since the grid-scale of regional models is of the order of a kilometer or more, dm_{ambout}/dt should be of the order of dm_{ambin}/dt well within an emission grid cell. The downwind flux equilibration distance depends on local surface conditions and wind speeds. This may have some implications for the minimum spatial resolution of a dust transport model. The distance could be varied as a function of the soil and vegetative cover if good geographic data bases of such information are available.
- The analysis above was for the case with the wind perpendicular to the road. For the wind at an angle to but not parallel to the road, the analysis would be almost the same, but with a wider "effective road width." However, for an infinitely long road parallel to the wind the assumptions of the analysis are inapplicable.

3.3 Information Gaps

Information gaps include:

- Flux measurements through the ceiling above the road, dm_{ceil}/dt.
- Measurement of the vertical profile with distance downwind of the source from the ground to the top of the control volume.
- Geographic data bases of soil type, physical characteristics, vegetative cover, and morphology that are needed to employ the concepts on a regional basis.
- Deposition velocities for the wide variety of obstructions around different dust suspension areas.

3.4 Recommendations

Testing of the above semiempirical model should be done in a relatively clean environment where fugitive dust from vehicular traffic is a dominant source of PM_{10}. Such testing could best be done in arid or semi-arid locations.

The semiempirical model should be upgraded with more complex and representative submodels of deposition and diffusion near the surface.

4. FUGITIVE DUST EMISSION FACTORS NEED TO BE APPROPRIATE (FINDING #3)

Many of the past fugitive dust emission inventories were inaccurate since they used inappropriate emission factors. Emission factors should be based on physical models rather than statistically significant variables and should be consistent for different source types with similar suspension mechanisms (e.g., mechanically reentrained dust from paved and unpaved roads; wind erosion from desert, agricultural, and vacant land as well as unpaved roads and construction sites). There have been recent advancements in characterizing the factors that make up the empirically derived emission factor equations as well as the reformulation of the emission factor equations that need to be taken into account. For example, improved algorithms for fugitive dust emissions from construction operations are currently being implemented. Also, the unpaved road dust algorithm is currently under review. Some categories may still be inaccurate because re-evaluations have not been undertaken. Categories needing more work are dust from unpaved roads and wind erosion. There are inconsistencies in the way different regulatory agencies calculate windblown dust emissions; scientists must agree on an approach that considers the physics of soil suspension by high winds.

4.1 Overview of Emission Factors

Emission factors and emission inventories have long been fundamental tools for air quality management. Emission estimates are important for developing emission control strategies, determining applicability of permitting and control programs, ascertaining the effects of sources and appropriate mitigation strategies, and a number of other related applications by an array of users including federal, state, and local agencies, consultants, and industry. Data from source-specific emission tests or continuous emission monitors are usually preferred for estimating a source's emissions because those data provide the best representation of the tested source's emissions. However, test data from individual sources are not always available and, even then, they may not reflect the variability of actual emissions over time. Thus, emission factors are frequently the best or only method available for estimating emissions, in spite of their limitations. Many emission factors are contained in AP-42, a compendium of emission factors that is compiled and updated periodically by the EPA.

4.1.1 What Is An Emission Factor?

An emission factor is a representative value intended to relate the quantity of a pollutant released into the atmosphere with an activity that causes the release. These factors are usually expressed as the weight of pollutant divided by a unit weight, volume, distance, or duration of the activity emitting the pollutant (e.g., kilograms of particulate emitted per hectare of soil disturbed). These factors are typically averages of available data that are assumed to represent long-term averages for many specific emitters within a source category (i.e., a population average).

The general equation for emission estimation is:

$$E = A \times EF \times (1 - ER/100) \qquad (4\text{-}1)$$

where E is the emissions rate, A is an activity level, EF is the emission factor, and ER is the overall percentage emission reduction efficiency for controlled emissions. When estimating emissions for a long time period (e.g., one year), the emission reduction efficiency (ER) should account for upset periods as well as routine operations.

The extent of completeness and detail of the emissions information in AP-42 (and other emission factor references) is determined by the information available from published references. Emissions from some categories are better documented than others. For example, only a few areas of the country have been tested for emissions from agricultural operations while emission tests of unpaved roads have been completed in many locations. It is necessary to look at the documentation for an emission factor to determine if it is applicable to the specific situation. The fact that an emission factor for a pollutant or process is not available from EPA or other sources does not imply that the source does not emit that pollutant or that the source should not be inventoried.

4.1.2 Uses Of Emission Factors

Fugitive dust emission factors in AP-42 are neither EPA-recommended emission limits (e.g., best available control technology [BACT], or lowest achievable emission rate [LAER]) nor standards (e.g., National Emission Standard for Hazardous Air Pollutants [NESHAP] or New Source Performance Standards [NSPS]). Use of these factors as source category-specific permit limits and/or as emission regulation compliance determinations is not recommended by EPA. Because emission factors represent an average of a range of emission rates, approximately half of the applications will have emission rates greater than that predicted by the emission factor and the other half will have emission rates less than that predicted by the factor. As such, a permit limit using an AP-42 emission factor would result in half of the sources being in noncompliance.

Also, for some fugitive dust sources, emission factors may be presented for operations having air pollution control techniques in place. Factors noted as being influenced by control technology do not necessarily reflect the best available or state-of-the-art controls, but rather reflect the level of typical control for which data were available at the time the information was published. Sources are usually tested more frequently when they are new and when they are believed to be operating properly, and either situation may bias the results.

Source-specific tests can determine the actual pollutant contribution from an existing source better than can emission factors. Even then, the results will be applicable only to the conditions existing at the time of the testing. To provide the best estimate of longer-term (e.g., yearly or typical day) emissions, these conditions should represent a source's routine operations.

4.1.3 Variability of Emissions

Average emissions differ significantly from source to source and, therefore, emission factors frequently may not provide adequate estimates of the average emissions for a specific source. The extent of between-source variability that exists, even among similar individual sources, can be large depending on the source type. Although the causes of this variability are considered in emission factor development, this type of information is seldom included in emission test reports used to develop AP-42 factors. As a result, some emission factors are derived from tests that may vary by an order of magnitude or more. Even when the major process variables are accounted for, the emission factors developed may be the result of averaging source tests that differ by factors of five or more.

Air pollution control techniques also may cause differing emission characteristics. The type and volume of traffic, road cleaning techniques, the use of different roadbed materials for unpaved roads, soil type and moisture all influence the amount of materials suspended. In the case of

windblown dust, the reservoir effect, where the loose materials are removed initially, can limit the duration of a wind event. This reservoir effect is also a factor in paved road emissions. Often, the nature of these variable parameters are not fully documented in the emission test reports (at least not in a form conducive to detailed analysis of how varying parameters can affect emissions) and therefore may not be accounted for in the resulting emission factors.

Before applying AP-42 emission factors to estimate emissions from new or proposed sources, or to make other source-specific emission assessments, the user should review the latest literature and technology to be aware of circumstances that might cause such sources to exhibit emission characteristics different from those of other, typical existing sources. The subject source type and conditions represent those source(s) analyzed to produce the emission factor.

Emission factors are developed to represent long-term average emissions, so testing is usually conducted at normal operating conditions. Parameters that can cause short-term fluctuations in emissions are generally avoided in testing and are not taken into account in test evaluation. Thus, using emission factors to estimate short-term emissions will cause even greater uncertainty. Short-term emissions from a single specific source often vary significantly with time (i.e., within-source variability) because of fluctuations in process operating conditions, ambient conditions, and other such factors. To assess within-source variability and the range of short-term emissions from a source, one needs either a number of tests performed over an extended period of time or continuous monitoring data from an individual source.

4.1.4 Methodology for Deriving Emission Factors

Emission factors are derived from experiments on specific sources that are believed to represent the general behavior of a source category. Most of the EPA emissions factors (USEPA, 1999) are based on horizontal fluxes from an emitting area such as a road or vacant lot (Cowherd and Englehart, 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. This is most easily accomplished along roadways or open areas that have consistent traffic and confined boundaries.

Each of the downwind samplers is used to represent the amount of dust that is 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.

Vertical flux is an alternative to horizontal flux as a method to estimate fugitive dust emissions (Gillette, 1977). Vertical flux is proportional to particle density, surface friction velocity, and the difference between particle concentrations at different elevations above ground level. Only a fraction of the horizontal flux moves upward, depending on meteorological, aerosol, and surface variables. These variables can be obtained with a configuration similar to that of Figure 4.1 (Nickling and Houser, 1999).

Currently used emission factors based on integrated horizontal flux adequately represent suspendable dust, the amount that leaves a dust-generating surface. These factors do not adequately represent transportable dust, the fraction of suspendable dust that is likely to travel more than a few hundred meters from the emitter. Future revisions to AP-42 may consider factors for horizontal fluxes above certain elevations that can be associated with appropriate zones of influence. For example, fluxes below 2 meters agl would not be used to estimate impacts at distances larger than 1 to 10 km from the source, but would be available to estimate impacts on nearby receptors. Vertical flux estimates are more appropriate for determining fugitive dust impacts over urban and larger spatial scales.

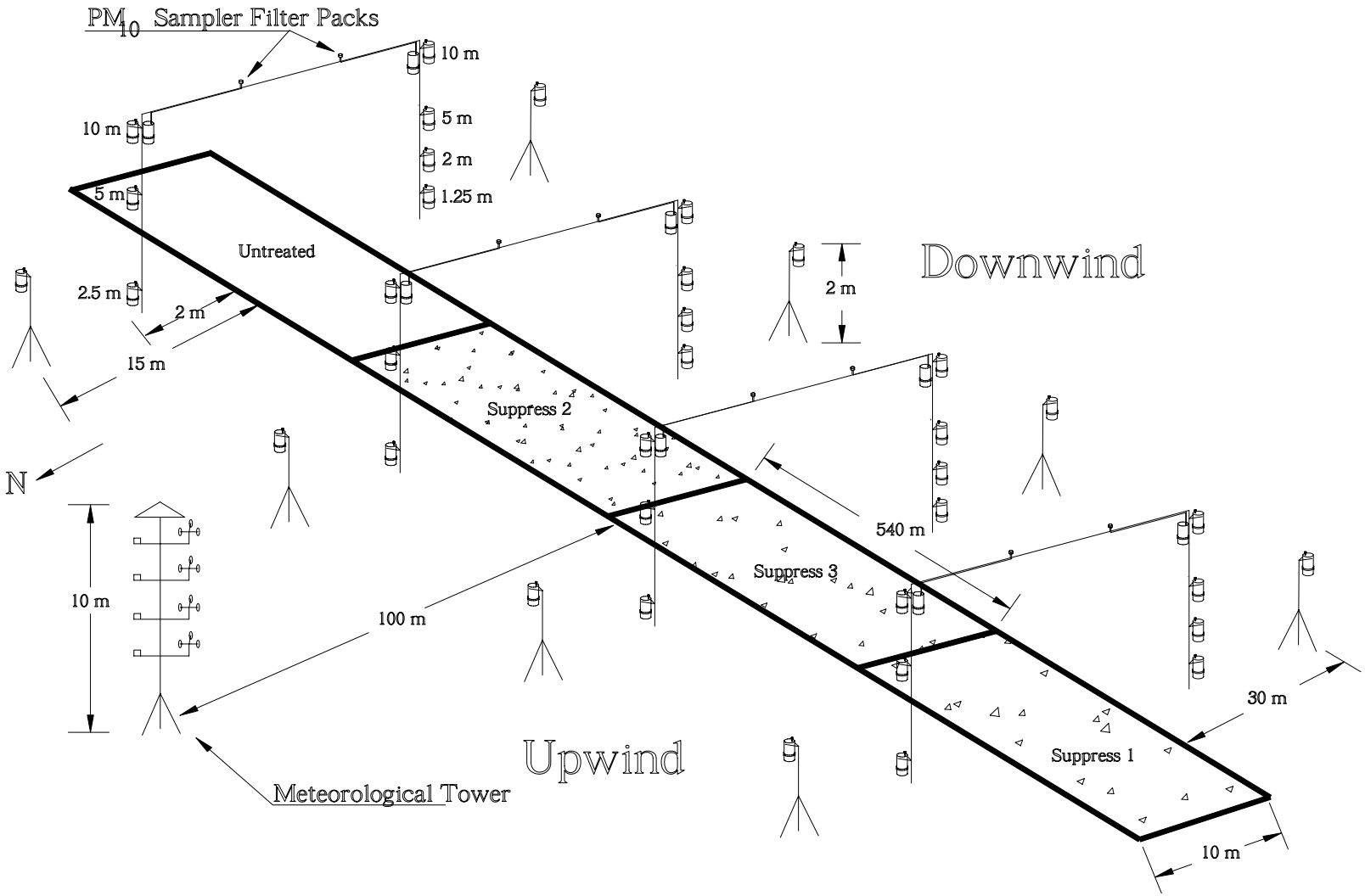
Estimation of fugitive dust emissions is not an exact science. Fugitive dust emissions depend on particle sizes, surface loadings, surface conditions, wind speeds, atmospheric and surface moisture, and dust-suspending activities (e.g., vehicle and equipment usage and operating characteristics) and these parameters are not readily obtainable when compiling inventories for large geographic areas. Fugitive dust emissions estimates contain a high amount of variability owing to the difficulty in obtaining estimates of these meteorological and physical variables, which often vary widely on a national or smaller regional basis. Both systematic (bias) and random errors are likely present for both spatially- and temporally-averaged emission rates.

4.2 Wind Erosion

Wind erosion and windblown dust are of interest to both air quality and agricultural scientists. The former are interested in the emission and transport of wind-generated dust, as total suspended particulate (TSP), particulate matter smaller than 10 µm (PM_{10}), or particulate matter smaller than 2.5 µm ($PM_{2.5}$). Agricultural scientists, on the other hand, are interested in wind erosion – the soil movement and soil loss from a field. This material is larger in size (50 to 100 µm or larger) and is transported horizontally from the field edge, rather than vertically into the atmosphere. Thus, although they share similarities, there are some fundamental differences in the approaches to modeling wind erosion and windblown dust, since the former is concerned with horizontal transport, while the latter is concerned with vertical transport.

Most modeling approaches treat both horizontal wind erosion and vertical dust emissions as processes that do not take place until some threshold wind velocity or threshold friction velocity occurs. To estimate vertical particulate emissions during a windblown dust event it is first necessary to obtain both the large scale and small scale wind fields. Once a threshold wind velocity has been exceeded, the wind erosion (horizontal flux) is related to the difference between the observed wind speed and the threshold value, multiplied by the wind speed raised to some power (2, 3, or 4). Then, because of the processes involved in suspension and transport of aerosol particles, it is necessary to estimate horizontal movement of sand or soil. A detailed discussion of the processes involved in wind erosion are presented below.

FIGURE 4-1. Example of horizontal flux measurement system around an unpaved road (Watson et. al., 1996)



4.2.1 Processes That Affect Suspension And Transport Of Aerosol Particles

Particle suspension begins with soil grain motion, whether it is due to wind erosion, or mechanical processes such as vehicles driven over unpaved surfaces. Studies on wind erosion were initiated after the Dust Bowl years of the 1930s, with several investigators focusing on the drag and lift forces acting on grains of soil that initiated airborne soil transport (e.g., Bagnold, 1941; Chepil 1945, 1951). Initial motion requires aerodynamic forces to exceed adhesive forces. Once soils are put into motion, three processes are involved: surface creep, saltation, suspension.

Creep is the phenomenon whereby soil particles larger than ~1000 µm and soil aggregates that are too large to be picked up by wind move along the ground because of wind forces and impact with other soil grains. In the process of *saltation*, some grains are lifted by the air but fall back to the surface after a short distance. Saltation can be thought of as the horizontal flux of soil across the surface. These are the particles that are the most damaging to crops. Saltating particles also lead to suspension of smaller particles including PM_{10}. *Suspension* occurs when grains smaller than ~20-30 µm become entrained in turbulent air motion and are transported longer distances. The vertical flux of dust corresponds to the mass of particles suspended.

Researchers classify the parameters that lead to wind erosion in different manners; however, the following characteristics appear in some form or other in most wind erosion studies:

- Wind energy – the shear stress on the soil surface due to the wind
- Soil erodibility – the vulnerability or resistance of soil to erosion by wind
- Surface cover – the amount of biomass on the field
- Surface roughness – the soil microtopography
- Soil moisture and crusting
- Erodible field length in the direction of the wind

Soil erodibility is determined by a number of soil properties, including aggregate size distribution, density and stability; bulk density; crust thickness; amount of loose erodible material; and the relative area covered by crust and nonerodible and erodible material (Zobeck, 1991). The surface roughness is related to the size of soil aggregates. As aggregate stability decreases, the size of soil aggregates decreases, the geometric mean diameter decreases, and the erodible fraction increases. Both climate and field treatments affect the erodibility of the soil by affecting aggregate stability. Drought increases the likelihood for wind erosion because soil moisture decreases, vegetative surface cover decreases, and aggregate stability decreases, with a resulting increase in erodible fraction (Merrill et. al., 1997). Merrill and coworkers studied the effects of tillage treatments and the effects of periods of drought by examining aggregate size distributions versus time for spring wheat-fallow cropping systems. That work showed tillage effects on aggregate stability, with the lowest aggregate stability for the conventional tillage treatment, and the highest aggregate stability observed for no-till fields. Periods of drought were found to have an even larger impact on aggregate stability.

The above parameters have been studied largely with respect to their effects on wind erosion of soil, i.e., the horizontal transport process. The efficiency of the vertical dust release process, characterized by the ratio of the vertical (suspension) to horizontal (saltation) fluxes, also varies from soil type to soil type. Sabre et. al. (1997) noted that this ratio depends upon the wind

friction velocity and its threshold value, as well as the kinetic energy of saltating particles striking the soil surface. The wind friction velocity, u_*, is related to the shear stress at the surface, τ_o, through the identity:

$$u_* \equiv \left(\frac{\tau_o}{\rho}\right)^{1/2} \tag{4–2}$$

where ρ is the air density. In practice, the friction velocity can be calculated using the logarithmic wind profile equation:

$$u_z = \frac{u_*}{k} \ln \frac{z}{z_o} \tag{4–3}$$

where u_z is the wind velocity, z is the height above the surface at which the wind velocity measurement is taken, k is the dimensionless von Karman constant (0.4), and z_o is the aerodynamic roughness height of zero average wind velocity. The threshold friction velocity is the minimum wind stress needed to overcome the adhesive forces that hold soil particles in place (Gillette, 1988) and is a function of surface characteristics as well as chemical characteristics, such as carbonate or organic matter content. Large differences exist between threshold friction velocities for desert sands and those for agricultural soils due to differences in precipitation, surface treatment, soluble salts that may be present on desert soils, vegetative cover, or land use (Gillette, 1988). Threshold friction velocities for various soil samples have been catalogued for the National Resource Inventory (NRI) national soil inventory.

4.2.2 Technical Support of Finding #3 *vis a vis* Windblown Dust

Windblown dust quantities in emissions inventories are inaccurate for several reasons. These include using wind erosion models developed for purposes other than air quality emission inventories; temporally- and spatially-averaged meteorological data that average out wind gusts; inadequately characterized soil and topographic characteristics; and models that do not properly characterize the emission processes. In developing the various wind erosion and dust emission models, a variety of experimental techniques have been developed. These experimental methods are presented in the remainder of this section, followed by a discussion of various models for estimating dust emissions from wind erosion.

4.2.2.1 Methodologies for Estimating Dust Emission Rates From Wind Erosion

Most of the research on wind erosion from agricultural areas has focused on the *horizontal* transport of dust (saltation component), because it is the saltation process that causes the most damage to crops, and results in soil losses due to wind erosion. In order to quantify wind erosion, a number of aeolian dust samplers have been designed. Horizontal dust fluxes have been measured using the Big Spring Number Eight (BSNE) sampler, the modified Wilson and Cooke (MWAC) sampler, the suspended sediment trap (SUSTRA), and the wedge dust flux gauge (WDFG). Goossens and Offer (2000) have recently examined the performance of these aeolian dust samplers using wind tunnel experiments and field calibrations, and the literature describing various samplers is extensive; for example, Rubow (1995) provided a detailed overview of commercial samplers that are still available and in use.

Early studies attempted to estimate *vertical* dust flux (suspension component) by treating it as some fraction of the total horizontal mass flux at the downwind perimeter of an eroding area. However, Gillette and Passi (1988) pointed out that fields with a large fetch may come to a saturation erosion level, such that the horizontal flux does not change with increasing the length of the field. Since the vertical dust flux does not reach a similar saturation level, estimates of vertical dust fluxes that are determined from a constant ratio of the total soil loss from the field may be in error. Still, most windblown dust models treat the suspension component as being related to the saltating component.

Wind Tunnel Experiments. Experimental methods for estimating vertical dust emission rates due to wind erosion have included both wind tunnel and field studies. A number of wind tunnel studies on both wind erosion and on the resuspension of PM have been carried out. Saxton and co-workers in Washington State designed a portable wind tunnel (Pietersma et. al., 1996) for measuring the relative susceptibility to erosion for a variety of soils for the Columbia Plateau. Field trials were conducted for various surface roughnesses and residues (Horning et. al., 1996). The results from these experiments have been used to characterize soils in eastern Washington for the effect of surface residue and soil surface roughness on wind erosion, and the relative erodibility of soils. In these studies, the wind tunnel was co-located at instrumented field sites for calibration of these erodibility factors (Saxton et. al., 2000). A difficulty associated with the use of wind tunnels for determining the relationship between wind speed and either wind erosion or dust emission is the apparent role of saltation. Shao et. al. (1993) examined the role of saltation bombardment in the suspension of PM in a portable wind tunnel and concluded that saltation, as opposed to direct aerodynamic lift, was the dominant mechanism maintaining dust emission fluxes. To achieve fully developed saltation requires a sufficient amount of fetch, which is not available in a wind tunnel experiment. In several experiments, the saltating particles were introduced artificially. However, it is not clear whether the saltation phenomenon thus achieved mimics that which occurs naturally.

Field Measurements. There are two approaches for measuring dust emissions in field-scale experiments: vertical profiling, and upwind-downwind measurements. In vertical profiling, the K-theory of turbulence is applied. This approach has been utilized for the flux measurements of a number of gaseous species in the absence of fast chemical sensors. The method is based upon the assumption that the turbulent flux is proportional to the vertical concentration gradient:

$$F = -K \frac{\Delta c}{\Delta z} \qquad (4-4)$$

where F is the flux, K is a turbulent diffusion coefficient, c is the concentration, and z is the elevation. The coefficient K is a function of atmospheric stability, and its specification may be achieved via a variety of methods, all of which have limitations. Nevertheless, in the absence of fast chemical sensors, that would otherwise permit more accurate flux measurement techniques, the K-theory approach continues to be used for estimation of vertical fluxes, and has been frequently applied to estimate fugitive dust fluxes (e.g., Gillette, 1977; Nickling and Gillies, 1993; Sabre et. al., 1997; Saxton et. al., 2000).

As an example of this approach, in the form used by Saxton et. al. (2000), the vertical dust flux is determined by making an analogy between mass flux and momentum flux and assuming a

logarithmic wind profile. The resulting equation relates the dust flux to particulate concentration gradient, the wind velocity difference, and the friction velocity:

$$F = \frac{-ku_*(c_2 - c_1)}{\ln\left(\frac{z_2}{z_1}\right)} \qquad (4-5)$$

where c_2 and c_1 are the mean dust concentrations at heights z_2 and z_1, u_* is the friction velocity, and k is the dimensionless von Karman constant (0.4). It should be noted that this equation only applies to times during which the friction velocity is at or above the threshold friction velocity.

Beyond the difficulties associated with determination of the friction velocity and application of K-theory, it should be noted that this type of experiment provides an estimate of the dust emission flux at a single point in space. In the upwind-downwind type of experiment, the emission estimate is made over an integrated area. In this experimental approach, upwind, downwind, and far-downwind particulate concentrations are measured, along with meteorological conditions, and source contributions are calculated using a dispersion model. The benefits of using such an approach include the determination of emissions over a field-scale, and being able to conduct the experiments without interfering with the activity being studied. The disadvantages include the uncertainties associated with dispersion modeling, which can introduce errors of a factor of two or more (Turner, 1994). This approach has been used to measure fugitive PM_{10} emissions from selected agricultural practices in the San Joaquin Valley (e.g., Matsumara et. al., 1992). In their work, Matsumara and coworkers measured aerosol concentrations both upwind and downwind of the source. The upwind concentrations were subtracted from the downwind concentrations, and the resulting "net" concentrations were compared to calculated concentrations from a model that included dispersion modeling from a line source and particle settling velocities. The concentrations calculated using dispersion models are subject to uncertainties associated with several parameters, including the wind speed, since Gaussian-type dispersion models are based upon the assumption of vertically constant winds. The effective height of release (emission height) also depends on wind speed, and so there are also uncertainties associated with its designation. Dispersion parameters are estimated using Pasquill stabilities, which consider discrete classes, while in reality stability is a continuum.

The uncertainties introduced with the dispersion modeling associated with upwind-downwind types of experiments may be reduced by the use of atmospheric tracers. By using an inert tracer such as sulfur hexafluoride (SF_6) and releasing it in a way that mimics the emission source of interest (e.g., line source for imitating roads, point source for stack emissions), the need for dispersion modeling can be eliminated. By measuring the upwind and downwind concentrations of both the inert tracer and the air pollutant of interest, and knowing the emission rate of the inert tracer, the emission rate of the pollutant can be calculated from the ratio of its concentration to that of the inert tracer, multiplied by the inert tracer emission rate (Claiborn et. al., 1995). Even if the inert tracer is not released in a manner similar to the emission source of interest, the release of the inert tracer provides a way to determine the dispersion parameters for use in a dispersion model interpretation of upwind-downwind pollutant concentrations (Kantamaneni et. al., 1996). The use of SF_6 as an atmospheric tracer for PM_{10} over short distances has been demonstrated for

determining emission factors from both paved and unpaved roads (e.g., Claiborn et. al., 1995; Kantamaneni et. al., 1996). We are not aware of any studies that have attempted to use an inert tracer such as SF_6 to determine PM_{10} emissions from agricultural fields, or in wind erosion studies.

Scale-up. In all of the above emission measurement approaches, the problem of scale-up to the regional scale can be significant. Surface inhomogeneities coupled with the nonlinear influences in the wind erosion and dust emission processes can lead to significant uncertainties in the use of small-scale measurements for regional-scale modeling (Moore et. al., 1993). These uncertainties have been addressed by the research community by either attempting to incorporate the detail needed to fully characterize these effects, or by lumping them into an overall proportionality coefficient, or by some combination of these approaches. For example, in an attempt to reconcile dispersion modeling results that incorporated these emission measurements with downwind concentrations, Claiborn et. al. (1998) compared PM_{10} concentration predictions to actual measurements for two wind events and found that a "calibration factor" was necessary to match predictions with observations. On the other hand, once the calibration factor was applied, the temporal and spatial trends were consistent with observations, and the resulting model has been used to examine the effects of potential changes in farming practices on downwind air quality.

4.2.2.2 Methodologies for Modeling Dust Emission Rates From Wind Erosion

Wind erosion models for agricultural soils fall into two categories: those that predict wind erosion (horizontal soil flux) and those that predict suspended dust emissions due to wind erosion (vertical dust fluxes). The early wind erosion equation and its revision (WEQ, RWEQ) both address horizontal fluxes. The most recent model developed by USDA scientists, the Wind Erosion Prediction System (WEPS), predicts both horizontal soil flux and vertical dust flux. Because most dust emission models relate vertical dust flux to horizontal soil flux, both wind erosion and windblown dust models are discussed below.

Horizontal Soil Loss Prediction Models

Wind Erosion Equation (WEQ) The early results of Bagnold (1941) and Chepil (1945, 1951) were incorporated into an empirical wind erosion equation (Woodruff and Siddoway, 1965) used by the U.S. Soil Conservation Service (now the Natural Resources Conservation Servicer [NRCS]) and other agencies throughout the country to inventory the potential of soils for wind erosion (Hagen, 1991). The resulting wind erosion equation (WEQ) is based upon empirically derived relationships between the major factors controlling wind erosion. The WEQ estimates the average annual mass of soil transported off the downwind edge of the field. It does not estimate PM_{10} emissions. The equation expressed in functional form is:

$$\text{Wind Erosion} = f(I, K, C, L', V') \tag{4.6}$$

where Wind Erosion is the potential average annual soil loss, I is the soil erodibility index, K is the soil ridge roughness factor, C is the climate factor, L' is the unsheltered distance across a field, and V' is the equivalent vegetative cover.

The EPA modified the WEQ to estimate windblown dust emissions (USEPA, 1974):

$$E_s = a\ I\ C\ K\ L'\ V' \tag{4.7}$$

where E_s is the suspended particulate emission factor in tons/acre/year, and a is intended to account for the fraction of the eroded soil that becomes total suspended particulate matter (TSP), estimated to be 0.025 for agricultural lands, and 0.038 for unpaved roads. To estimate PM_{10} emissions, the California Air Resources Board used a factor to account for the PM_{10} content of TSP (Air Resources Board, 1997):

$$E_s = a\ k\ I\ C\ K\ L'\ V' \tag{4.8}$$

where $k = PM_{10}/TSP = 0.5$.

Although not developed for air quality applications, there have been further attempts to use the WEQ to produce agricultural windblown dust emission estimates specifically for California. Because the WEQ was developed based upon data from the Midwestern US, the WEQ does not take into account many of the environmental conditions and farming practices in California. The California Air Resources Board staff modified the WEQ to improve the emission estimate for California, especially to better reflect seasonal changes. The modified equation, the ARBWEQ1, represents a major improvement with respect to both the annual emissions estimate and the monthly emissions profiles as suggested by ambient monitoring source apportionment results, as well as crop production patterns (Francis et. al., 1997).

WEQ is currently the most widely used method for assessing average annual soil loss by wind from agricultural fields. The primary user of WEQ is the NRCS. When the development of WEQ was initiated (~40 years ago), it was necessary to make it a simple mathematical expression, readily solvable with the computational tools available. However, WEQ has fundamental weaknesses because of its empirical representation of erosion processes which have lead to a number of efforts to improve the accuracy, ease of application, and range of WEQ.

Revised Wind Erosion Equation (RWEQ) Fryrear et. al. (1998) added improvements to the original wind erosion equation, and validated their contributions with field data. A significant modification included in the RWEQ is the assumption that the maximum transport capacity occurs a short distance downwind from the field boundary, beyond which no net soil loss occurs. The RWEQ has been tested against approximately 50 site-years worth of wind erosion data. As with the original WEQ, the RWEQ does not predict PM concentrations or vertical dust emissions. Moreover, since the RWEQ assumes a maximum transport capacity for saltation, it would be inappropriate to scale vertical dust fluxes to the horizontal saltation fluxes predicted with this model.

Vertical Dust Flux Prediction Models

EPA Model (AP-42) The PM_{10} emission factor equation for wind generated particulate emissions from mixtures of erodible and nonerodible surface material subject to disturbance listed in AP-42 is written as follows (USEPA, 1999):

$$E = 0.5\sum_{i=1}^{N} P_i \tag{4.9}$$

where E is the PM_{10} emission factor (g/m^2 per year), N is the number of disturbances per year (where disturbance is defined as an action that results in the exposure of fresh surface material) and P_i (g/m^2) is defined as the erosion potential corresponding to the observed (or probable) fastest mile of wind for the ith period between disturbances (where the fastest mile represents the wind speed corresponding to the whole mile of wind movement that has passed by the one mile contact anemometer in the least amount of time).

The total erosion potential for a dry exposed surface subject to disturbances is calculated for each wind erosion event (i.e., periods when the friction velocity exceeds the threshold friction velocity) using the following equation taken from AP-42:

$$P = 58(u_* - u_{*t})^2 + 25(u_* - u_{*t}) \tag{4.10}$$

where P is the erosion potential (g/m^2), u_* is the friction velocity (m/s), and u_{*t} is the threshold friction velocity (m/s). Note: $P = 0$ for $u_* \leq u_{*t}$.

AP-42 states that the resulting calculation is valid only for a time period as long or longer than the period between disturbances. The friction velocity (a measure of wind shear stress on the erodible surface) is determined from the following equation derived to explain the wind speed profile in the surface boundary layer for neutral stability conditions:

$$u_* = 0.4\, u_z / \ln (z/z_0) \quad (\text{for } z > z_0) \tag{4.11}$$

where u_z is the wind speed measured at height z above the test surface (m/s), z is the height above the test surface (cm), and z_0 is the roughness height (cm). The roughness height (a measure of the roughness of the exposed surface) is the height above the surface at which the wind speed would become zero if the logarithmic equation describing the wind speed profile were extended down to this height. Emissions generated by wind erosion are dependent on the frequency of disturbance of the erodible surface because each time that a surface is disturbed, its erosion potential is restored. Any natural crusting of the surface binds the erodible material and reduces the erosion potential. For uncrusted surfaces, AP-42 recommends estimating the threshold friction velocity from the mode of the dry aggregate size distribution of samples of the surface material (according to the sieving procedure described in AP-42) and using Table 13.2.5-1 in AP-42 (USEPA, 1999), or alternatively the graphical relationship described by Gillette (1980).

<u>*NAPAP* Model</u> Gillette and Passi (1988) developed a model to estimate total dust production by wind erosion for the US, for the National Acid Precipitation Assessment Program (NAPAP). The dust model they developed does not use existing estimates of soil erosion. During the development of dust emission models for NAPAP, Gillette and coworkers found that the dust emission flux from soils could be described in terms of a threshold friction velocity that incorporates the effects of soil type, soil moisture, soil texture, and vegetative cover. Gillette and Passi's dust model utilizes the following equation:

$$E = C\sum_{i=1}^{N} R_i g(L_i) A_i \Delta T \int_{u_{ti}}^{\infty} G(u) p_i(u) du \qquad (4-12)$$

where E is the mass of dust emitted per unit time ΔT, C is a calibration constant, i is the summation index over N erodible areas within a region, R_i is the soil roughness factor, $g(L_i)$ is the field length (L_i) factor, A_i is the surface area, G(u) is the vertical mass flux of dust as a function of wind speed (u), $p_i(u)$ is the probability density function of wind speed during the time period of interest, and u_{ti} is i^{th} threshold wind speed for dust emission.

The vertical dust flux function, G, incorporates the threshold wind speed as follows:

$$G = C_2 C_d^2 u^4 \left(1 - \frac{u_t}{u}\right) \qquad (4-13)$$

where C_2 is a constant, C_d is the drag coefficient, and u_t is the threshold wind speed above which erosion takes place. The probability density function of wind speed was characterized using a two-parameter Weibel distribution, and the threshold velocities, vegetative residue, soil roughness function, land area, and land use were all taken from the National Resources Inventory (NRI). Written a slightly different way, the algorithm of Gillette and Passi relates the dust flux to the difference between the wind velocity and the threshold wind velocity, multiplied by the wind velocity raised to the third power:

$$\text{Dust Flux} \sim u^3(u-u_t) \qquad (4\text{-}14)$$

This general approach – correlating the dust emission to a threshold wind speed or threshold friction velocity – has been used repeatedly, but the form of the equation varies from user to user. For example, Nickling and Gilles (1993) obtained a similar equation from dust fluxes obtained from micrometeorological measurements. Tegen and Fung (1994) also used an earlier equation from Gillette (1978) that related the dust flux to the wind speed raised to the second power:

$$\text{Dust Flux} \sim u^2(u-u_t) \qquad (4\text{-}15)$$

The NAPAP methodology was modified for the purpose of assessing annual PM_{10} emissions for EPA's emissions trends report (Barnard and Stewart, 1992). Barnard and Stewart simplified the NAPAP model and used year-specific information to produce annual emissions estimates. Expected dust flux was estimated according to:

$$I = kC_2 C_d^2 \left(\frac{u^4}{0.886^4}\right) \Gamma(3, x) \qquad (4-16)$$

where I is the dust flux, k is a PM_{10} multiplier (0.9), C_2 is a constant (4×10^{-14}), C_d is the drag coefficient, u is the mean wind speed, Γ(3,x) is the incomplete gamma function, and x is a function of the wind speed and the threshold wind speed:

$$x = \left(u_t\left(0.886\right)/u\right)^2 \qquad (4-17)$$

In the above equation, the threshold wind speed was determined from the threshold friction velocity, which again is a function of soil type and precipitation. To calculate the emission flux due to wind erosion, requires information on the average monthly wind speed, total monthly precipitation, and anemometer height. The resulting model qualitatively predicts trends in emissions; however, there is considerable uncertainty associated with the magnitude of the emissions predicted. Uncertainties associated with this model result from the characterization of each state by a single soil type, the treatment of precipitation information, use of an average wind speed, and ignoring the significant variability in the threshold friction velocity. It should be noted that these emissions equations are very sensitive to both the threshold wind speed and the actual wind speed. The significance of the error introduced with the use of average wind speeds cannot be ignored. The effects of wind speed averaging have been shown to lead to significant under-prediction of dust emissions (Stetler and Saxton, 1997). At a minimum, when applying equations such as that of Gillette and Passi (1988), the user should be aware of the averaging time used for the wind speed and threshold wind speed, and use data with the same averaging time.

Models Predicting Both Horizontal Soil Flux and Vertical Dust Flux

Texas Erosion Analysis Model (TEAM) The Texas Erosion Analysis Model (TEAM), developed at the Wind Engineering Research Center at Texas Tech University (Singh et. al., 1997), predicts both wind erosion and dust emissions. Key parameters taken into account include wind profile, friction velocity, soil cover factor, variable threshold friction velocity, soil detachment function, maximum transport rate, eroding field length, soil erodibility, mechanics of dust generation, saltation height, reference zone concentration, dust concentration with height, and visibility prediction. The TEAM model treats the threshold friction velocity as a variable rather than a constant value, which is the approach taken in the WEQ and RWEQ. The threshold friction velocity equation in the TEAM model is related to the particle diameter and friction velocity (Bagnold, 1941). The equation also considers the effect of electrostatic bonding, which is important for small particles, and soil moisture.

The maximum transport rate is a function of kinetic energy, and is proportional to the friction velocity multiplied by the difference between the squares of the friction velocity and the threshold friction velocity:

$$\text{Horizontal Soil Flux} \sim u_*(Su_*^2-(u_{*t}/G)^2) \qquad (4\text{-}18)$$

where S is a surface cover factor, and G is a gustiness factor. In order to run the TEAM model, necessary input parameters include average hourly wind speed, relative humidity, clay content, D50 and D75 particle size, particle size distribution, surface cover factor, soil erodibility, soil bulk density, and length of the erosion segments from which the soil detachment and transport rate and dust generation are determined. The use of a gustiness factor along with hourly averaged wind speed is important. Farber et. al. (1996) found that the occurrence of wind gusts over 13.5 m/s at a10 meter height was a key ingredient to saltation. The TEAM model, which

also includes a visibility prediction module, was used to analyze a dust storm that caused a major traffic accident in California in 1991, with reasonable results.

Wind Erosion Prediction System (WEPS) The Wind Erosion Prediction System (WEPS) developed by Hagen and others (e.g., Hagen, 1995) is intended to replace the WEQ and RWEQ. The technical documentation for WEPS can be obtained from the WERU website, http://www.weru.ksu.edu/weps.html. WEPS simulates both wind erosion and the vertical flux of dust. Unlike the previous wind erosion models, WEPS is a process-based, continuous, daily time-step model that simulates weather, field conditions, and erosion. Saltation, creep, suspension, and the PM_{10} component of erosion can be separated out and examined individually. WEPS simulates complex field shapes and topography, and takes into account spatial and temporal variability of field conditions and soil loss or deposition. Because WEPS is a continuous, daily, time-step model, it also simulates the processes that modify a soil's susceptibility to wind erosion.

The structure of WEPS is modular and includes seven submodels**:**

- WEATHER – provides daily weather data. This submodel consists of statistical databases developed from historical weather records. There are two programs included in this submodel. WINDGEN stochastically models wind direction and subdaily wind speed. CLIGEN generates temperature, precipitation, solar radiation, and dew point temperature on a daily basis.
- HYDROLOGY – accounts for changes in temperature and water status of the soil. This submodel maintains a daily soil water balance by tracking the amount of daily precipitation, irrigation, snow melt and accumulation, runoff, evapotranspiration, and deep percolation.
- SOIL – simulates changes in the soil properties between management events. This submodel estimates temporal changes in soil crust properties including crust thickness, cover fraction, density and stability, ridge and furrow dike height, random roughness, aggregate stability, density and size distribution, and bulk density.
- CROP – accounts for crop growth. This submodel calculates daily production of biomass, including roots, leaves, stems, and reproductive organs, as well as leaf and stem areas. CROP is able to take into account temperature stress, water stress, and nutrient stress conditions. The CROP submodel has been tested against field data for soybean, winter wheat, corn, grain sorghum, and oats, with the correlation coefficient ranging from 0.72 to 0.87.
- DECOMPOSITION – simulates the decrease in crop residue biomass due to microbial activity. The process is modeled as a first order reaction, and uses temperature and moisture as inputs. Biomass residue after harvest is partitioned between standing, surface, buried, and root pools.
- MANAGEMENT – models step changes in the soil and biomass conditions generated from typical management practices such as tillage, planting, harvesting, and irrigation.
- EROSION – models the wind power on a subhourly basis. This submodel calculates friction velocities and threshold friction velocities in each subregion, computes soil loss and deposition, and updates the surface variables if changed by erosion, generates simulation region grid points and initializes values, updates global subregion variables, and outputs information to files. A series of coupled partial differential equations are solved by the finite difference method. This submodel solves conservation of mass for the creep, saltation, suspension, and PM_{10} components, separately.

The concept of the maximum transport capacity beyond which no net horizontal soil loss occurs (introduced in the RWEQ) is also incorporated into the EROSION submodel. The transport capacity is determined from the friction velocity (u_*) and threshold friction velocity (u_{*t}):

$$\text{Transport Capacity} = C_s u_*^2 (u_* - u_{*t}) \tag{4-19}$$

where C_s is the saltation transport parameter, with a value of approximately 0.4 kg s^2 m^{-4}. PM_{10} emissions are treated as resulting from direct emission of PM_{10}-size soil material, abrasion of saltation and creep particles, and suspension due to saltating particles. The direct PM_{10} emission component is limited by the transport capacity; however, abrasion and saltation are related to the horizontal saltation particle velocity and so do not reach a saturation value. The WEPS model tracks surface changes, biomass changes, and soil water content and utilizes statistical wind characterizations, and so requires a very significant amount of detailed information. We are not aware of any air quality studies in which the WEPS model has been utilized.

Columbia Plateau PM_{10} Project As part of the Columbia Plateau PM_{10} Project, Claiborn et. al. (1998) modeled windblown agricultural dust for the Columbia Plateau in eastern Washington State. Claiborn and coworkers calculated the vertical PM_{10} dust flux using the approach of Gillette and Passi (1988) substituting friction velocity and threshold friction velocity for wind speed and threshold wind speed, respectively. PM_{10} threshold friction velocities were estimated as a function of soil type and vegetative residue, and ranged from 0.4 m/s for the most erodible soils on irrigated bare or dryland fallow fields to 1.60 m/s for the least erodible soils in range lands. In the application of this model to two dust events, the hourly averaged wind speeds used as inputs were corrected for short-term gusts, in a manner similar to that utilized in the TEAM model (Singh et. al., 1997).

Currently, as the Columbia Plateau PM_{10} Project continues, another empirical model is being developed to predict the contribution of dust emissions from wind erosion of upwind agricultural fields to regional PM_{10} concentrations. The modeling approach taken by Saxton et. al. (2000) is a two-step model, in which the horizontal flux of eroded soils is first calculated, then a corresponding vertical flux of PM_{10} is calculated. The empirical equation for predicting the horizontal flux of eroded soil is similar in form to the original WEQ, but applied on an event basis. The horizontal flux is envisioned to be a function of the event wind energy, soil erodibility, vegetative surface cover, soil surface roughness, and soil surface wetting and crusting. A constant threshold wind velocity for initiation of erosion is assumed. The horizontal flux of soil is proportional to the square of the wind speed multiplied by the difference between the wind speed and the threshold wind velocity:

$$\text{Horizontal Soil Flux} \sim u^2(u - u_t) \tag{4-20}$$

The vertical flux of suspended particles is related to the horizontal flux through the wind profile characteristics, and the PM_{10} flux is related to the vertical flux of suspended particles through a soil dustiness index, which is the ratio of the mass of suspended PM_{10} particles collected using a tapered element oscillating microbalance (TEOM) to the mass of the suspended soil sample. The resulting relationship between the vertical dust flux and the wind velocity has the same form as

that used by Gillette and Passi (1988), as well as by Claiborn et. al. (1998). The emission model of Saxton et. al. (2000) incorporates land use and soil type data specific to the agricultural areas of the Columbia Plateau, and has been incorporated into a regional GIS-based transport and dispersion model. The entire modeling system has been used to simulate five historical windblown dust events and has demonstrated its usefulness as a tool for evaluating potential control strategies.

Saudi Arabia Study In a very recent study, Draxler et. al. (2001) modeled PM_{10} emissions during desert dust storms in Iraq, Kuwait, and Saudi Arabia using a Lagrangian transport and dispersion model. The form of the emission flux equation followed that of Marticorena et. al. (1997), in relating the vertical dust flux to the difference in the squares of the friction velocity and threshold friction velocity:

$$\text{Vertical Dust Flux} \sim u_* (u_*^2 - u_{*t}^2) \tag{4-21}$$

In the algorithm used by Draxler et. al. (2001), a proportionality constant relates the surface soil texture to the PM_{10} dust emissions, and is defined as the ratio of vertical flux of PM_{10} to total aeolian horizontal mass flux. The threshold friction velocity is a function of surface roughness, which in turn is correlated to soil properties. The approach used by Draxler and coworkers to model PM_{10} emissions caused by wind erosion and based upon the emission flux equation of Marticorena and Bergametti (1995) followed a three-step process.

Step I. Obtain large scale and small scale wind fields. This step is largely meteorological and includes simulation or correct calculation of the large scale wind field from meteorological variables of pressure and temperature. Of greatest and most critical interest to dust emissions modeling is calculation of friction velocities and aerodynamic roughness heights for surfaces before any sand is transported or dust is emitted. These parameters are necessary in order to estimate the threshold friction velocity, and their calculation requires knowledge of smaller scale roughness. Information on the roughness must be translated into aerodynamic roughness heights and associated drag coefficients. The square-root of the drag coefficient when multiplied by the wind speed for neutral stratification gives the friction velocity. This work can be quite complex because roughness like vegetation can be inhomogeneous and non-randomly distributed. This produces local areas of high surface stress which for some kinds of surface sediments become local dust sources. These dust sources may exist even in conditions where average surface cover measured in terms of percentage of area of biomass per unit area would predict the surfaces to be stable. One must also deal with small hot spots of emissions imbedded within larger areas of depositional surfaces.

Step II. Prediction of sand movement (horizontal flux of particles $\geq$50 μm). Prediction of sand movement (saltation) is necessary for dust emissions modeling. The dominant wind erosion mechanism by which dust is emitted is sandblasting, which dislodges and propels fine particles by bombardment of sand grains. Evidence for the dominance of the sandblasting mechanism includes the following observations:

- The threshold velocity for direct aerodynamic entrainment of PM_{10} is much larger than that for particles 75 to 100 μm in size. However, PM_{10} particles are emitted for wind velocities lower than the threshold velocity for PM_{10}.
- The chemical composition of PM_{10} is the same as aggregated coatings of PM_{10}-size particles on sand-sized particles. PM_{10} particles are usually not found in non-aggregated form in the soil.
- Wind tunnel experimentation has shown that wind blowing over particles all smaller than 10 μm gives very short-lived dust emissions. However, when loose sand grains or other saltating particles are part of the surface sediment, sustained PM_{10} emissions are observed.
- Vertical PM_{10} fluxes and the horizontal flux of saltating sand are proportional to each other and to the friction velocity raised to the third power.
- Theoretical work shows that PM_{10} flux should be proportional to the friction velocity raised to the third power, and inversely proportional to the bonding energy of soil (Gillette, personal communication).
- Changes of the size distribution of PM_{10} have been shown to be related to changes of sandblasting kinetic energies.

The horizontal flux of sand may be modeled as:

$$q = A\frac{\rho}{g}u_*\left(u_*^{\ 2} - u_{*_t}^{\ 2}\right) \qquad (4-22)$$

where ρ is air density, g is acceleration of gravity, A is a dimensionless constant, u_* is friction velocity, and u_{*t} is threshold friction velocity required for initiation of sand movement by the wind. Marticorena and Bergametti (1995) calculated threshold friction velocity u_{*t} for unvegetated, uncrusted soils from knowledge of surface roughness and size distribution of loose particles on the surface. The calculation of the threshold friction velocity is given as:

$$u_{*_t} = \frac{u_{*ts}}{f_{eff}} \qquad (4-23)$$

where u_{*ts} is the threshold friction velocity for a smooth surface, and f_{eff} is the efficient friction velocity ratio defined as the ratio of local to total friction velocity:

$$f_{eff} = 1 - \left[\frac{\ln\left(\frac{z_o}{z_{ons}}\right)}{\ln\left[0.35\left(\frac{10}{z_{ons}}\right)^{0.8}\right]}\right] \qquad (4-24)$$

where z_o is the roughness height. The roughness length of the soil without any roughness elements z_{ons} was estimated by Greeley and Iversen (1985) to be $D_p/30$, where D_p is the mean soil particle diameter.

The value of the dimensionless constant A is not constant if there is wetting followed by crusting of the surface sediments or if there is a depletion of loose particles on the surface for a "supply-limited" surface. For example, a three year study conducted by Gillette and Chen (2000) showed

that the value of A ranged from a maximum of ~3.5 when the surface was covered with a loose sand to ~0 when the surface was a smooth crust with a few loose particles larger than 1 mm. A median value for supply-unlimited cases for another location was 2.8 (Gillette et. al., 1996).

Finally, the sand itself influences the drag coefficient C_{DS} after the threshold velocity u_t has been exceeded:

$$\sqrt{C_{DS}} = \sqrt{C_{DNS}} + 0.003\left(1 - \frac{u_t}{u}\right) \qquad (4-25)$$

where the drag coefficient for no sand movement C_{DNS} is defined by:

$$u_* = \sqrt{C_{DNS}}\,u \qquad (4-26)$$

<u>Step III. Suspended dust emissions.</u> Suspended dust is proportional to saltation or sandblasting as follows:

$$F_a = Kq \qquad (4-27)$$

where F_a is the vertical flux of dust, K is a dimensional constant with units of m^{-1}, and q is the horizontal flux of saltating particles. The value of K is not precisely known, but data sets of F_a versus q are available so that estimates of K can be made for certain soils. For one such data set, 94% of the surface soil samples were classified as having a "sand" texture based on the US Department of Agriculture's definition. For these samples the mean value for F_a/q was equal to 2×10^{-4} m^{-1} (Gillette et. al., 1997).

Draxler et. al. (2001) used their model to predict PM_{10} concentrations for locations in Kuwait. Their predicted concentrations were in fair agreement with measured concentrations. In addition, their modeling system predicted approximately the correct number of dust events over Kuwait, and agreed quantitatively with measurements at several locations in Saudi Arabia and Kuwait.

Comparison of Windblown Dust Prediction Models

Several windblown dust algorithms are compared, below. An earlier algorithm developed by Gillette (1978) was employed by Tegen and Fung (1994), who calculated the dust flux from the wind velocity and threshold wind velocity:

$$F = C_{TF}u^2(u - u_t) \qquad (4-28)$$

where F is the dust flux in $\mu g\ m^{-2}\ s^{-1}$, u is wind velocity in $m\ s^{-1}$, u_t is threshold wind velocity in $m\ s^{-1}$, and the dust constant, C_{TF}, ranges from 43 $\mu g\ s^2\ m^{-5}$ for undisturbed soils to 179 $\mu g\ s^2\ m^{-5}$ for disturbed soils. Tegen and Fung (1994) used this equation to estimate global dust emissions, and so this equation was developed for use with a 6-hour time step and much larger spatial grid size than that used by Claiborn et. al. (1998).

The various forms of emission equations used to predict the flux of windblown dust are summarized in Table 4-1.

Table 4-1. Forms of equations used to predict flux of windblown dust

Equation Form	References
Horizontal Soil Flux ~ $u_*^2(u_*-u_{*t})$	Saxton et. al., 2000
Horizontal Soil Flux ~ $u_*(Su_*^2-(u_{*t}/G)^2)$	TEAM model[1] Singh et. al., 1997
Vertical Dust Flux ~ $u_*^3(u_*-u_{*t})$	Gillette and Passi, 1988 Claiborn et. al., 1998 Saxton et. al., 2000
Vertical Dust Flux ~ $u^2(u-u_t)$	Gillette, 1978 Tegen and Fung, 1994[2]
Vertical Dust Flux ~ $u_*(u_*^2-u_{*t}^2)$	Draxler et. al., 2001

1. The TEAM model includes a gustiness factor (G) to correct the hourly averaged wind speed.
2. The form of the equation employed by Tegen and Fung was based upon wind velocities rather than friction velocities. The two velocities are proportional to each other and are related through the logarithmic wind profile equation, as discussed below.

Claiborn et. al. (1998) used the following form to model two dust events:

$$F = Cu_*^3 a_g \left(u_* - u_{*t}\right) \qquad (4-29)$$

where F is the flux of PM_{10} in μg m^{-2} s^{-1}, a_g is a constant to correct for the use of hourly averaged winds compared to the nonlinear effect of near-instantaneous gusts upon dust production (~1.20), u_* is the friction velocity corresponding to a 10-m wind velocity measurement (m s^{-1}), u_{*t} is the threshold friction velocity (m s^{-1}), and C is an empirical dust constant. For the early-season event of September 11, 1993, C = 9.6 x 10^{-3} g s^3 m^3. For the late-season event of November 3, 1993, C = 1.5 x 10^{-3} g s^3 m^3. This equation requires hourly friction velocities at 10 m, and is applied for 1-km grids.

The equation provided by Nickling and Gillies (1993) correlates the dust flux to the friction velocity raised to a power of 4.38:

$$F = 1.93 \times 10^3 u_*^{4.38} \qquad (4-30)$$

where the dust flux F is in μg m^{-2} s^{-1}, and u_* is in m s^{-1}. The equation provided by Draxler et. al., (2001) is based upon a 6-hour time step and was developed for desert sands:

$$F = KA\left(\frac{\rho}{g}\right)u_*\left(u_*^2 - u_{*t}^2\right) \qquad (4-31)$$

where the dust flux F is in g m^{-2} s^{-1}, K has a value of approximately 2 x 10^{-4} m^{-1}, A ranges from 0 to 3.5 with a mean value of 2.8, and the ratio of air density (ρ) to gravity (g) is 120.8 g s^2 m^4.

Threshold wind velocities, or threshold friction velocities, must be determined, and are tabulated in numerous sources. The friction velocity for a neutrally stable atmosphere (which is applicable

for windy conditions) can be determined in a couple of ways. From the logarithmic wind speed profile for neutral conditions, the wind velocity is related to the friction velocity as follows:

$$u = \frac{u_*}{k} \ln\left(\frac{z-d}{z_o}\right) \qquad (4-32)$$

where k is the von Karman constant (~0.4), z_o is the aerodynamic roughness length or the height where the wind speed goes to zero, and d is the displacement height or approximately 0.7 times the height of the vegetation canopy. For bare ground, d = 0. The friction velocity and the aerodynamic roughness length can be determined experimentally over a variety of surfaces by measuring the wind speed at two heights.

Figure 4-2 compares the dust flux calculated from each of the algorithms listed in Table 4-1, assuming a threshold friction velocity of 0.4 m s^{-1}. There is a significant difference in the estimated fluxes from these various algorithms. The difference between the CP3 curves for the November and September 1993 events is due to the significant difference in the fragility of the soil early in the season, since the soil is recently disturbed in September. The differences between some of these curves is related to the time step assumed and the grid size.

4.2.3 Limitations

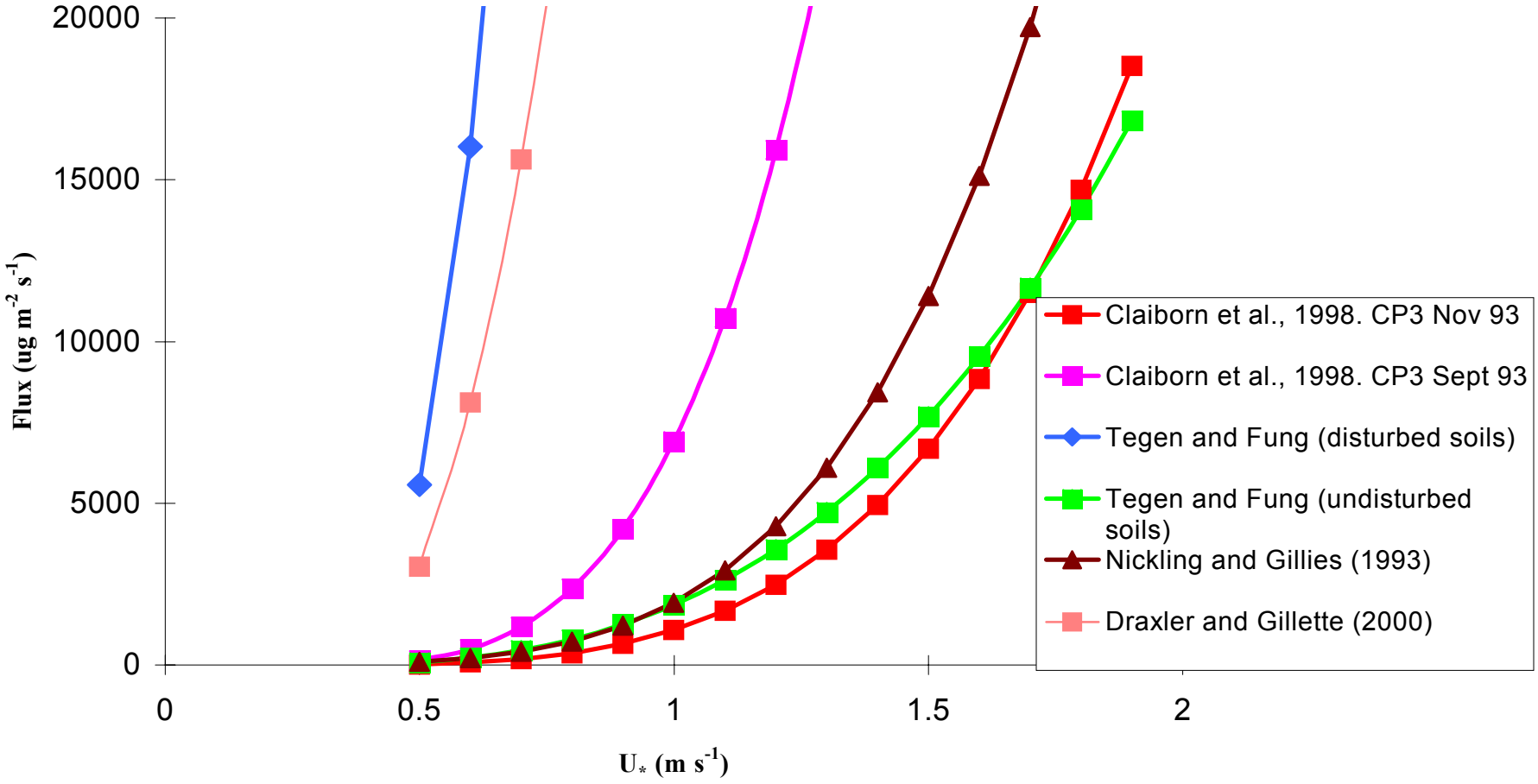
The emissions models discussed above are subject to a number of limitations. These are related to the treatment of horizontal versus vertical flux, assumption of constant threshold friction velocity, averaging of wind velocities so that gustiness is not taken into account, complexities associated with adequate treatment of surface inhomogeneities, and use of forms of emission equations that do not represent physical processes.

Horizontal versus vertical fluxes and maximum transport capacity. While it has been shown that horizontal soil losses reach a saturation value (i.e., maximum transport capacity), vertical dust fluxes do not. On the other hand, saltating particles are usually required to provide the energy needed to suspend particles, so that the suspension component is proportional to the saltation component (Gillette et. al., 1995). Caution should be exercised when developing emission inventories so that emissions are not calculated as a fraction of the total saltation component leaving the downwind edge of the field.

Constant versus variable threshold friction velocity. Specification of an appropriate threshold friction velocity for use in any of the emission models discussed is especially important because the wind velocity must exceed the threshold value before any emissions are predicted to occur. If the specified threshold velocity is too high, then low wind speed dust events will not be predicted. On the other hand, many of the studies discussed above demonstrated that the threshold velocity is highly variable and is a function of surface conditions. It is thus likely that inappropriate specification of threshold velocities represents a limitation in modeling windblown dust emissions in many cases.

Use of average wind velocities. The importance of wind gusts has been discussed. The use of average wind velocities, both in time and in space, tends to smooth out the extremes of the gusts.

Figure 4-2. Vertical dust flux predicted as a function of friction velocity; assumed a threshold friction velocity of 0.4 m/s.



Since all of the emission equations shown are proportional to wind velocity raised to some power greater than one, the error introduced with averaging will be magnified.

Incorporation of small-scale topographic features and variability in input parameters. Small-scale topographic features will be highly variable and will impact surface characteristics that in turn will affect both the threshold friction velocity and the local friction velocity. All of the emission equations discussed therefore will be highly sensitive to topographic features.

Use of physically consistent methodology. Prediction of sand or soil movement is necessary for dust emissions modeling because the dominant wind erosion mechanism by which dust is emitted is sandblasting by saltating particles. The threshold velocity for direct aerodynamic entrainment of PM_{10} is much larger than that for particles 75 to 100 μm in size, yet PM_{10} particles are emitted for wind velocities lower than the threshold velocity for PM_{10}. The vertical flux of PM_{10} mass is proportional to the horizontal flux of saltating particles, and theoretical work has shown that the PM_{10} flux is proportional to friction velocity raised to the third power, and inversely proportional to the bonding energy of soil.

4.2.4 Information Gaps

Many of the gaps in the information needed to calculate windblown dust emission factors were discussed in the previous section. The most thorough modeling attempt to date is the USDA-developed WEPS. While the WEPS approach takes into account many of the processes that affect the wind erodibility of a field, as well as suspended dust emissions during a windblown dust event, the WEPS model requires a very large amount of input data. Moreover, it has not yet been demonstrated for use in predicting PM_{10} concentrations during windblown dust episodes.

Prediction of vertical dust fluxes during windblown dust events is sensitive to a number of parameters, including field characteristics and topography. Specification of the appropriate threshold friction velocity may represent one of the most important parameters. The role of wind gustiness has been examined but only briefly, yet it may also be a significant parameter when time-averaged wind speeds are used. This is to be expected, since windblown dust typically contributes to an emission inventory on an episodic rather than continual basis.

The theoretical aspects of saltation and suspension are currently under development and it is not clear which, if any, of the models mentioned here is the more accurate.

4.2.5 Recommendations

The WEPS model developed by USDA scientists shows great potential for modeling not only wind erosion but also windblown dust. However, the WEPS model has not yet been demonstrated for prediction of PM_{10} emissions for use in air quality models or in emission inventories. The feasibility of using the WEPS model for predicting PM_{10} emissions requires testing. It is not clear how well WEPS will perform in these applications, nor is it apparent how user-friendly the system will be.

Adopting a modeling approach such as that outlined by Draxler et. al. (2001) will provide a reasonable compromise between incorporating too much detail and not enough. The method of Draxler et. al. requires a limited amount of *a priori* information on the surface. The threshold friction velocity is estimated, and the horizontal transport of soil is calculated. The dust emission

flux is proportional to the horizontal transport through a proportionality coefficient. This proportionality constant is needed for specific soils. An examination of archived data sets that include both horizontal soil fluxes and vertical dust fluxes would allow a start at cataloging this proportionality constant for various soils.

4.3 Paved and Unpaved Roads

4.3.1 Technical Support of Finding #3 *vis a vis* Mechanically Resuspended Road Dust

Unpaved Roads Little new work has been performed to evaluate emission factors from either paved or unpaved roads. A revised version of the AP-42 emission factor for unpaved roads was published in September 1998 (U.S. EPA, 1999). For the September 1998 revision, results from 180 PM_{10} and 92 TSP field tests were used to develop the unpaved road emission factor. The current AP-42 PM_{10} emission factor for unpaved roads is:

$$E = 2.6\ (Silt/12)^{0.8}\ (Weight/3)^{0.4}\ /\ (Moisture/0.2)^{0.3} \qquad (4\text{-}33)$$

where E is the PM_{10} unpaved road dust emission factor for all vehicle classes combined (pounds/vehicle mile traveled), Silt is the silt content (i.e., material less than 75 μm in diameter of the surface material (% mass), Weight is the average weight of the vehicle fleet (tons), and Moisture is the surface moisture content (%). The earlier version of the AP-42 emission factor included variables for speed and number of wheels contacting the unpaved road surface, but these were not found to be statistically significant in the re-analysis of the test data. EPA provides ranges of average silt loadings for different source types from 5.1% to 24%, with individual sample silt loadings ranging from 0.1% to 68%. The equation above overestimates average emissions for vehicle speeds lower than 15 mph. For speeds below 15 mph, EPA recommends multiplying the above equation by S/15, where S is the average vehicle speed (mph). The EPA lists the $PM_{2.5}/PM_{10}$ ratio as 0.15.

A control effectiveness modifier due to rain is often appended to the unpaved road emission factor equation to estimate annual averages:

$$P_{annual,county} = PrecipDays/365 \qquad (4\text{-}34)$$

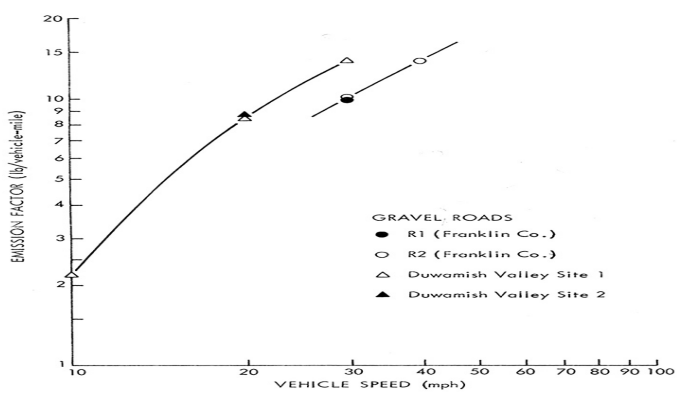
where PrecipDays is the number of precipitation days per year with at least 0.01 inches of rain as determined by the National Climatic Data Center. These data are usually sufficient to obtain spatial resolution for different counties within a state. This modifier assumes a 100% emissions reduction effectiveness for each day with at least 0.01 inches of precipitation. It should be pointed out that many scientists have questioned the validity of this assumption for arid areas of the western US.

Most of the current large-scale (e.g., regional or national) emission inventories have been developed using the earlier version of the AP-42 emission factor containing parameters for vehicle speed and number of wheels (Barnard, 1997; Barnard et. al., 1990, 1992; USEPA, 1998). The revisions to this earlier version of the AP-42 unpaved road emission factor (Cowherd et. al., 1974; Midwest Research Institute, 1983) were typically iterative modifications of the original factor first published in 1974, rather than new or alternative approaches to the formulation of the emission factor (Cowherd et. al., 1976, 1978). This is important for a number of reasons. It

could mean that the original formulation of the emission factor might have been flawed. Thus iterative improvements based on the original formulation would have built upon a potentially flawed formulation.

Figure 4-3, which is taken from the document detailing the original development of the AP-42 unpaved road emission factor, provides data on unpaved road emission factor measurements for two sites in Kansas along with data from a separate study conducted in the Duwamish Valley, WA by Roberts et. al. (1975). Text on page 40 of that document (Cowherd et. al., 1974) indicates that: *"The test results reported above indicate the total dust emissions from unpaved roads increase in proportion to the average vehicle speed, in the speed range of 30 to 40 mph. As shown in Figure 10* [Ed. Note: Figure 4-3 below], *this dependence is corroborated by the results of the Duwamish Valley study. Sehmel's data (Sehmel, 1973) on resuspension of tracer dust from asphalt roads indicates that the linear dependence extends up to 50 mph* [Ed. Note: Sehmel's data is not shown in Figure 4-3]. *Below 30 mph, however, both Duwamish Valley study and Sehmel's measurements indicate that emissions increase in proportion to the square of the vehicle speed. Since the typical speed range on unpaved roads is 30-50 mph, the linear dependence of dust emissions on vehicle speed was used in developing the correction factor."*

Figure 4-3. Emission factor/vehicle speed relationship from Cowherd et. al. (1974).



There are several important items to examine in Figure 4-3. First, the data were only collected at two vehicle speeds for the measurements made in the study conducted by Cowherd et. al. (1974). There were replicate samples evaluated at 30 mph, but still only two speeds (30 and 40 mph) were evaluated. Thus, any relationship developed between emissions and vehicle speed would necessarily be linear since there are really only two points to evaluate. Second, the information is plotted in log/log format. The Duwamish Valley data (representing three speeds for the study conducted by Roberts et. al., 1975) shows basically a linear relationship, when plotted in log/log format. A linear plot in log/log format indicates a power relationship between the variables. Thus, the dependence between emissions and vehicle speed would be an increase in emissions as some power of the vehicle speed rather than linearly with vehicle speed. Examination of the original data for both the Duwamish Valley and Sehmel's data on tracer resuspension on an asphalt road both indicate that the relationship between emissions and vehicle speed is a power of the speed rather than linear.

Figure 4-4 shows the original data from the Duwamish Valley study by Roberts et. al. (1975). This figure indicates that the relationship between emissions and vehicle speed is non-linear. Figure 4-5, is reproduced from Sehmel's study (Sehmel, 1973). The data are again plotted in a log/log format yielding roughly linear relationships, again indicating emissions are related to speed via a power function. Sehmel confirms this power function when he states: *"The resuspension rate increases with vehicle speed. Of particular interest is that the resuspension rate increases with the square of car speed when the car is driven through the tracer."*

Figure 4-4. Relationship between emissions and vehicle speed from Duwamish Valley study (Roberts et. al., 1975).

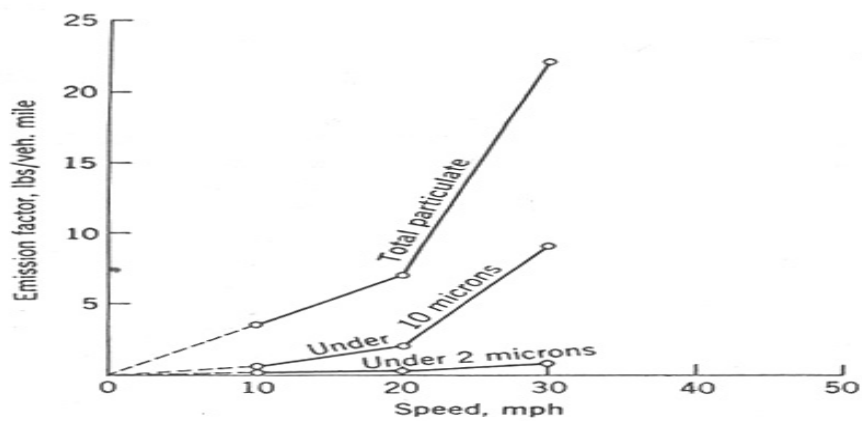
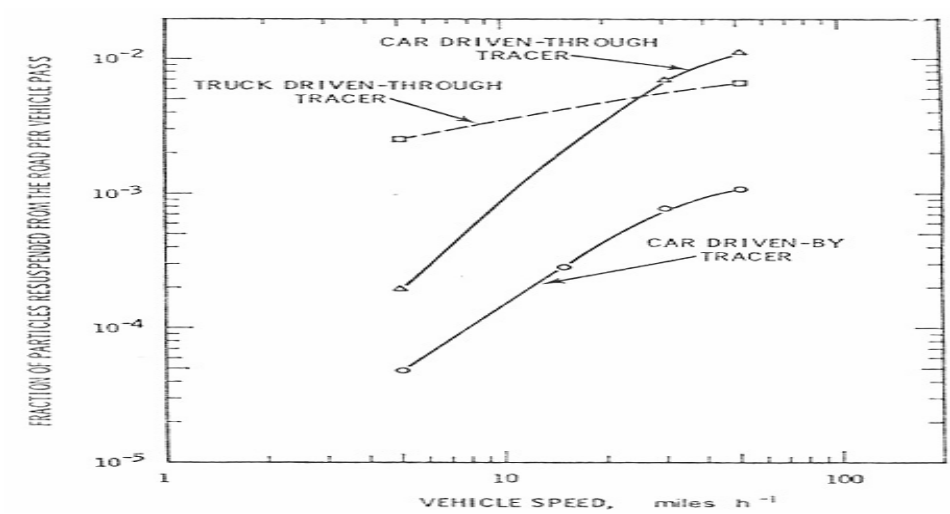


Figure 4-5. Relationship between particle resuspension and vehicle speed from Sehmel's (1973) resuspension study.



As indicated above, the current version of the AP-42 emission factor does not utilize vehicle speed or number of wheels in the calculation algorithm. Instead, surface moisture content has

been substituted for the number of dry days, and vehicle speed and number of wheels have been discarded as parameters in calculating emissions since these parameters were not found to be statistically significant. During the data evaluation phase resulting in revision of the emission factor, vehicle speed was omitted since it provided no additional improvement in the overall correlation. This provides an example of one potential flaw in the use of statistical evaluations to determine real world parameterizations of physical processes. Intuitively, it is apparent that when the speed of a vehicle increases on an unpaved road, the visual plume also increases, and thus emissions must also be increasing. Thus removing vehicle speed from the emission factor equation simply because it provides no additional information in a statistical correlation analysis makes no physical sense. Addition of a moisture term (while creating problems related to source activity determinations) makes physical sense because surface moisture certainly affects the emissions from an unpaved road.

Alternative forms of emission factors for unpaved roads have been proposed by a number of researchers, but little additional work to verify these factors has been performed (Barnard et. al., 1987, 1988; Becker and Takle, 1979; Claiborn et. al., 1995; Dyck and Stukel, 1976; McCaldin and Heidel, 1978; Moosmuller, et. al., 1998; Pinnick et. al., 1985; Williams et. al., 1988, 1989).

Paved Roads For paved roads, the formulation for the AP-42 emission factor (USEPA, 1999) has changed little since it was originally developed (Cowherd, et. al., 1977). The current AP-42 emission factor (updated in October 1997) was developed based on the results from regression analysis on data from 65 paved road dust tests that measured PM_{10} (USEPA, 1999). The current PM_{10} emission factor is:

$$E = 0.016\ (\text{Silt Loading}/2)^{0.65}\ (\text{Weight}/3)^{1.5} \qquad (4\text{-}35)$$

where E is the PM_{10} paved road dust emission factor for all vehicle classes combined (pounds per vehicle mile traveled), the factor 0.16 recommended by EPA represents a ratio of PM_{10}/TSP of 0.20, Silt Loading is the silt loading of the surface material (g/m^2), and Weight is the average weight of all vehicle types combined (tons). This equation is qualified for the range of variables that encompasses the tests from which it was derived. These are silt loadings between 0.02 and 400 g/m^2, mean vehicle weights of 2 to 42 tons, and mean vehicle speeds of 10 to 55 mph. Default silt loadings for normal roads are 0.1 g/m^2 for roads with at least 5,000 vehicles per day and 0.4 g/m^2 for roads with less than 5,000 vehicles per day. For dirty roads, such as those with visible carryout or road sand on them, the default values are 0.5 g/m^2 for roads with at least 5,000 vehicles per day and 3 g/m^2 for roads with fewer than 5,000 vehicles per day. These defaults may differ widely from actual values; great improvements in emissions estimates are gained with a few simple silt loading measurements. The EPA lists the $PM_{2.5}/PM_{10}$ ratio as 0.25.

For paved roads, the new field studies that have been performed have utilized tracer studies (Claiborn et. al., 1994, 1995; Kantamaneni et. al., 1996) and/or evaluations that produced single valued emission factors without corrective parameters other than traffic volume (Wittorff et. al., 1994; Venkatram et. al., 1999). The results of these recent tracer studies have produced values which are significantly higher than the AP-42 emission factor for low traffic volume roads (i.e., less than 10,000 vehicles per day) and lower than AP-42 emission factors for high traffic volumes (i.e., greater than 10,000 vehicles per day). Venkatram et. al. (1999) showed no clear

correlation between measured emission factors and AP-42, nor a consistent pattern of emissions either higher or lower than AP-42 based on traffic volume. A study conducted by Hurrell and Zimmer (1994) in Denver produced a PM_{10} emission factor that was related directly to traffic volume as a power function. With the exception of expressways, the emissions were significantly lower than those predicted by AP-42. More recent work by Venkatram (2000) has shown that the AP-42 emission factor model yields highly uncertain estimates because (1) it lacks a mechanistic basis, (2) its formulation is highly dependent on the data set used to derive it, and (3) the accuracy of the model is completely determined by the methods used to measure emissions.

Additional work has also been carried out to further refine the silt loading term. An historic analysis of collected values for silt loading was performed in the update to the AP-42 paved road chapter completed in October 1997. The results of that analysis suggested that silt loading values, especially for high volume roads, should be significantly lower than those previously published in AP-42, and that for the limited number of samples available for analysis, the values for limited access roads (e.g., freeways) were reasonably similar throughout the country.

4.3.2 Limitations

With the exception of extremely localized inventories, the current forms of the emission factors for paved and unpaved roads are difficult to utilize for large-scale (e.g., regional or national) inventories. This limitation is largely related to the correction parameters (silt loading, silt content, moisture content, etc.). Databases of information for these parameters at a county or sub-county scale across broad regions are simply not available. By necessity, large-scale emission inventories must use values interpolated from or equal to adjacent measured values, or default values presented in AP-42 to determine emissions. This leads to a reduction in the overall accuracy in the emissions estimates.

4.3.3 Information Gaps

What is not clear from the information presented above is why there has not been a convergence between the paved and unpaved road emission factors. The only common correction parameter between the two is the weight term (albeit with a different exponent in the two equations). With the exception of the replenishment of the particle reservoir available for resuspension, one would expect that the other processes responsible for emissions would be similar. To date, no studies have evaluated whether it is possible to produce a single equation that could effectively predict emissions from both paved and unpaved roads.

An additional information gap relates to determination of the fine fraction (e.g., $PM_{2.5}$) of paved and unpaved road emissions. EPA has recently initiated efforts to revise the information related to the fine fraction of both paved and unpaved road emissions. The majority of particle size information developed for these sources was based on using a single sampler at one height on the exposure profiler tower. Thus, no empirical data exists to determine the vertical distribution of fine particles being emitted from road surfaces. The recent evaluation of the fine particulate fraction ratios by EPA was initiated due to recognition that there were particle bounce problems associated with the samplers used in the earlier studies. The particle bounce problems led to over estimation of the fine particle mass caused by larger particles bouncing through the impactor and being deposited on the final filter stage. The final filter stage typically represented a particle size in those studies of approximately 2 μm.

Virtually no data currently exist related to the particle reservoir available from these sources. Exhaustion and replenishment times for the particle reservoir remain virtually undocumented.

4.3.4 Recommendations

In recent years, EPA has made efforts to improve the data used in the "correction" parameters of the AP-42 emission factor equations. For example, additional information on silt content of unpaved roads has been added and a reexamination of the particle size fraction information has been performed. The new AP-42 unpaved road emission factor now adds a moisture term, since moisture certainly plays a role in the overall emissions from this source. More modest data improvements have been made to the paved road emission factor. Detailed suggestions for improvement for all $PM_{2.5}$ emission sources were developed by Barnard et. al. (1995). These suggestions included developing $PM_{2.5}$ emission factors for on-road vehicles, obtaining information on the temporal variation in activity levels, and quantifying the efficiency of different control measures.

The bottom line is that while there are certainly potential problems with the AP-42 emission factors, they remain (at least for now) the primary source for providing emission estimates for paved and unpaved roads. Further work to evaluate the forms of the equations and additional work to look at other studies that may provide additional information in order to revise and improve the AP-42 factors, such as those performed by the Illinois State Water Survey (ISWS) as part of the NAPAP, should be performed. For example the ISWS work (Williams et. al., 1988) developed an unpaved road emission factor that included a term to account for the percent of millimeter-sized particles in the tire track. That equation explained 93 percent of the variance in the measured emission factor. Perhaps the millimeter-sized particle term represents a first order estimate of the potential reservoir of suspendable particles.

The work by ISWS and others also includes measurements made at multiple downwind distances. This information could be utilized to determine the decay rate of the horizontal flux, which could potentially be used to assess impaction removal mechanisms for small particles since they are unlikely to have been removed via gravitational settling over the distances measured (typically less than 100 meters). The vertical concentration distribution measured during exposure profiling studies could be evaluated to determine if the data are sufficient to make an assessment of the vertical flux. In addition, since exposure profiling studies normally have an upwind sampler (to correct for background concentration), for studies where exposure profiling is used, it should be possible to use the upwind/downwind method to calculate emission rates, which could be used as an additional validation methodology, although the dispersion model used to calculate the emission factor may require revision.

Despite the potential errors inherent in the AP-42 emission factors for paved and unpaved roads, the lack of a significant supporting body of analyses and minimal geographic coverage makes it difficult at this time to recommend alternative emission factors. The AP-42 emission factors for paved and unpaved roads have the longest history, the majority of associated data, and the highest utility of any emission factors that have been presented.

5. SOURCE ACTIVITY LEVELS NEED TO BE ACCURATE (FINDING #4)

Fugitive dust emission inventories have often used inappropriate estimates of the extent of the source activity levels. There is a large uncertainty in the extent of the reservoir of suspendable particles (which is especially true for wind erosion and paved roads), as well as the effects of meteorological variables and human intervention. Available activity databases need to be identified and evaluated with respect to utility and scale, especially for scales of local and regional source influence at western Class I areas. Emission factor development efforts should consider the availability of activity level data and produce factors normalized to the reported activities.

5.1 Technical Support of Finding #4

This finding applies not only to the extent of activity levels (which for paved and unpaved roads is vehicle miles of travel [VMT]) but also applies to the data used in the "correction" parameters in the emission factor equations (e.g. silt content, silt loading, average fleet vehicle weight, etc.). As the emission inventory scale goes from local to regional to continental, locating consistent, readily available databases of activity data or information required for the correction parameters becomes increasingly limited. The possible exception is VMT on paved roads. Since paved road mileage far exceeds unpaved road mileage, state or local transportation departments are more likely to collect and report VMT on paved roads. Information on individual field parameters for wind erosion or silt loading for specific roads or areas are much more difficult to locate. For example, the current National Emission Trends (NET) inventory (USEPA, 1998) provides emissions for all area sources for every county in the U.S. In order to correctly estimate emissions for unpaved roads in the NET (which uses the pre-1998 AP-42 emission factor equation discussed in section 4.3), county-level information would be required on unpaved road silt content, average fleet vehicle weight, average fleet vehicle speed, fleet average number of wheels per vehicle, days with precipitation greater than 0.01 inch, and VMT. There is no national database of this information available at a county-level; the majority of the information is not even available at a state-level. Thus a number of assumptions are required to produce county-level emission estimates.

Even when data are available, the utility is often limited. Table 5-1 provides the data available for VMT for local functional class unpaved roads from the Federal Highway Administration. Local functional class roads have the majority of travel on unpaved roads. The data in Table 5-1 indicate that only the mileage by average daily traffic volume (ADTV) category is available. Thus, unless EPA specifically contacted each state or county transportation department to obtain better data, these are the data used to determine VMT. VMT is determined from the data in Table 5-1 by assuming a default value for each ADTV category (e.g., 75 vehicles per day for Rural Gravel/Soil Surface roads for the 50-199 ADTV category), and then multiplying that default value by the number of days per year and by the number of miles of roadway in that ADTV category. Thus VMT for the Rural Gravel/Soil Surface for the 50-199 ADTV category would be 253,218,750. However, the value is actually somewhere between 168,812,500 and 671,873,750.

Information on silt content is even scarcer. Reasonable assumptions can be made for vehicle speed, vehicle weight, and number of wheels. The new AP-42 emission factor equation requires information on surface moisture content; however, a national database of unpaved road surface

moisture content does not exist. Substituting soil moisture levels may not be adequate since unpaved roads frequently are surfaced with a variety of gravel types (crushed limestone, washed river gravel) and the traffic on the roads may cause quicker drying than nearby soils.

Table 5-1. Miles of local functional class unpaved roads by ADTV category.

State	Rural Gravel/Soil Surface				Rural Unimproved Surface			
	<50	50-199	200-499	500+	<50	50-199	200-499	500+
Alabama	15,970	9,250	500	0	0	0	0	0

	Urban Gravel/Soil Surface				Urban Unimproved Surface			
	<200	200-499	500-2K	2000+	<200	200-499	500-2K	2000+
Alabama	357	243	239	27	0	0	0	0

Equivalent examples could be provided for paved roads or wind erosion. For example, to use the current EPA model for estimating wind erosion in the NET, information for each county is required for soil threshold friction velocity, monthly precipitation, average monthly wind speed, and soil acreage. Even with fairly extensive coverage, not every county has a meteorological measurement station within its boundaries. Thus, even the most commonly available data is frequently not available at the level required for the emission inventory effort.

5.2 Limitations

Local scale emission inventories, where data on activity levels or correction parameters used in the emission factor equations (or both) can be easily collected or obtained, are less likely to experience problems associated with inadequate or unavailable activity data. Once the emission inventory scale approaches a regional level, but is calculated at smaller sub-regional levels (e.g., multi-state but calculated at the county level), then serious problems obtaining activity data or correction parameters can arise. As a consequence, any larger-scale inventory prepared with emission factors that require correction parameters will have accuracy problems since gross assumptions about these parameters will be required.

Even in those cases where information is typically available for the activity data, the data that are available are frequently not of the correct geographic and/or temporal scale needed for the inventory purpose. Allocation factors are frequently required to create emission estimates at the geographic scale required. This can result in substantial uncertainty. A comparison between the NET inventory for paved and unpaved roads and a similar inventory prepared for California by the California Air Resources Board (CARB) found significant differences between county-level emissions prepared for each inventory. The main differences were primarily related to the allocation factor used to create county-level emissions in each inventory. The NET used allocation factors since county-specific activity data were not available nationally. CARB collected actual activity data from county transportation agencies. While the magnitude of the emissions were within a factor of two at the state level (which is generally regarded as good for fugitive sources), the actual county-to-county variability ranged from very small to several thousand percent.

5.3 Information Gaps

One of the greatest limitations associated with activity data is related to particle reservoir and the particle reservoir replenishment time period. Emission calculations (and the activity data used to support these emissions estimates) typically assume that the reservoir of particles available is infinite with no depletion. This is most certainly not the case. Additionally, the mechanisms leading to reservoir replenishment, the time period required for replenishment, and the effects of this replenishment process on the emission estimates have not been investigated.

Large scale databases of activity data and/or correction parameters that can be used for regional or larger scale emission inventories and that provide the ability to calculate emissions at a minimum of a county level for fugitive dust sources are largely unavailable. Some categories do have activity data that are suitable for use in estimating emissions. For example, VMT for paved roads is generally available. Some databases are adequate for emission estimates as long as trends in emissions are not required. For example, large scale soil characteristic databases such as the National Resource Inventory (NRI) soil inventory are available; however, these databases are not updated each year and thus are not truly sufficient for determining long term trends.

Information related to the long and short term effects of meteorological conditions (wind, precipitation) on emissions are not well documented. The exception is wind erosion, which by its nature deals with wind variability. However the effects of precipitation especially on paved and unpaved roads are not well documented. For example, brief periods of rain resulting in at least 0.01 inch are assumed to suppress emission completely when preparing an annual inventory. However, during a summer day, a brief shower that only slightly exceeds 0.01 inch of rain, followed by hot sunny conditions may result in dry conditions later in the day and subsequent fugitive dust emissions.

Precipitation can cause the creation of a crust on the soil surface which acts to lower wind erosion. Human activities and their relationship to emissions have also not been adequately studied. Activities such as dirt bike traffic in desert areas act to disturb the natural soil crust (frequently the result of precipitation and subsequent drying of the surface) and increase the likelihood of erosion.

5.4 Recommendations

If changes are not made to current emission factor formulations, significant improvements are required for the availability of activity data if large scale emission inventory accuracy is to be improved. The lack of large scale, consistent sets of information necessary to calculate emissions is probably the single largest contributor to emission inventory inaccuracy at the present time.

6. FUGITIVE DUST EMISSIONS ARE NOT CONTINUOUS PROCESSES (FINDING #5)

Emission inventories and the emission factors used to develop emission inventories incorrectly treat fugitive dust emissions as continuous processes.

6.1 Technical Support of Finding #5

Emission factors developed using the upwind/downwind method (Claiborn et. al., 1995; Dyck and Stukel, 1976; McCaldin and Heidel, 1978) are by default treating emissions as a continuous process, since a continuous line source dispersion model is used to back-calculate the emission factor. For paved roads, this assumption, especially for multi-lane, high average daily traffic volume (ADTV) is probably reasonable for most time periods. However, even for these roads, there are almost certainly time periods in the late evening/early morning where non-continuous situations exist. For unpaved roads, the assumption that it is a continuous line source is certainly not correct. This is evident from the data shown in Table 5-1. That table clearly shows that for Alabama rural gravel and soil surfaced roads, there are almost 16,000 miles of road with fewer than 50 vehicles per day. Thus, these roadways have fewer than three cars per hour on average, which would not qualify as a continuous line source. Emission factors determined using the exposure profiling method are less susceptible to this problem since they measure the mass flux from the road directly. In either case, the emission factors derived are primarily intended for longer averaging times (typically annual emissions) and thus may not adequately reflect the temporal and spatial averaging times of the ambient measurements or air quality models.

6.2 Limitations

There are two primary limitations for this finding. The first limitation is that the emission factors developed using dispersion models, that assume the source is a continuous line or area source, may not correctly characterize the emissions. The second limitation is that the emission factors developed for annual emission inventories may not correctly capture emission characteristics for non-annual time periods.

6.3 Information Gaps

Additional information is needed on the temporal aspects of the activities leading to emissions from these sources.

6.4 Recommendations

Some emission factors development efforts for fugitive sources have utilized dispersion modeling to back-calculate source strength (emission factors). When dispersion models have been used, they have assumed that the sources were operating continuously. Investigation into using “puff” type models is needed. Puff models assume that the emissions are instantaneous, rather than continuous. These models are more likely to adequately represent the emissions process. In addition, virtually all emission factor development has centered on developing emission factors that are primarily intended for annual emission estimates. Some types of fugitive emissions are seasonal in nature, or have a duration that is significantly shorter than annual or seasonal. Further work to characterize the emissions for shorter time frames is necessary.

7. ANNUAL FUGITIVE DUST EMISSION INVENTORIES ARE NOT SUFFICIENT (FINDING #6)

Annual emission inventories are not sufficient to develop emission control strategies for haze events. Improved seasonal and diurnal profiles are needed for use in emission inventories. Wind erosion is highly variable and poorly characterized. Many of these variations are affected by certain meteorological conditions that are not currently considered in emissions models.

7.1 Technical Support of Finding #6

Fugitive dust emissions are likely to change over annual, seasonal, daily, and even diurnal periods owing to differences in the material available for suspension and the activities that cause that suspension. Yearly and seasonal variations can be practically addressed in emission inventories. Day-to-day and diurnal emissions might be addressed for specific episodes for a few sources, but this requires prior planning to track emissions events which is impractical under most circumstances. Windblown dust has shown major annual average differences from year-to-year owing to the use of national wind statistics to estimate where and how often wind speeds exceed suspension thresholds and when rainfall would suppress emissions. Barnard and Stewart (1992) estimated agricultural wind emissions varying from 0.8 million tons per year in 1986 to 2 million tons per year in 1989. The addition of region-specific rainfall reduced emissions estimates by nearly an order of magnitude, and by different fractions during each year.

Campbell and Shimp (1998) combined land use with crop calendars and monthly wind speeds and precipitation to estimate fugitive dust emissions for each month throughout the year. Figure 7-1 compares PM_{10} and $PM_{2.5}$ emission estimates for the southern San Joaquin Valley of California for months representing different seasons. During the winter, there is little tilling or plowing and frequent storms keep the surface moist which result in lower emissions. During spring, fields have dried, some plowing and planting has commenced, but wind erosion is high owing to frequent, short-lived, dry frontal passages and dust emissions rise. During fall, winds are generally low, but there is much activity during the harvest which result in high emissions. These activities have been coupled with land use and soil characteristics in an ARC/INFO system that is undergoing continuing improvement in California (Shimp and Campbell, 1995). This method can also be extended to annual inventories for looking at annual trends.

Figure 7-2 shows how ambient PM_{10} concentrations from fugitive dust can vary throughout the day, in this case in response to changes in wind speed (Chow and Watson, 1997). The sampling location was selected to be within and typically downwind of a cluster of construction projects that graded substantial amounts of land. Wind speeds exceeding 4 m/s resulted in substantial increases in concentrations. Chow et. al. (1999) showed by spatially dense monitoring that these high values did not result from a single source, as the monitoring site was not located adjacent to any specific construction activity, but an aggregate of many contributions from different construction sites.

Figure 7-1. PM_{10} and $PM_{2.5}$ emissions estimates in the southern San Joaquin Valley for different seasons (Campbell and Shimp, 1998). The PM_{10} emissions are dominated by fugitive dust sources and correspond to areas of agricultural activity.

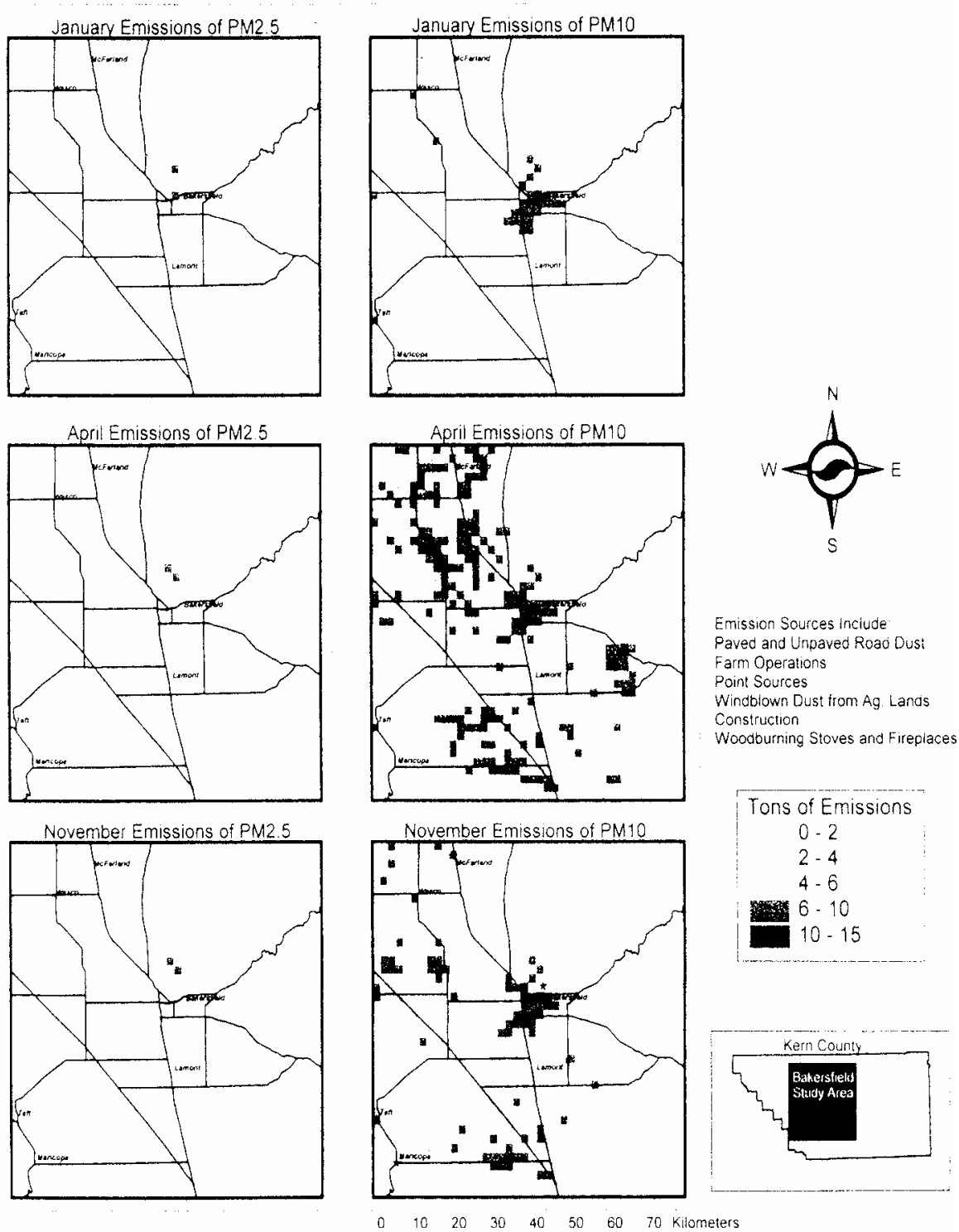
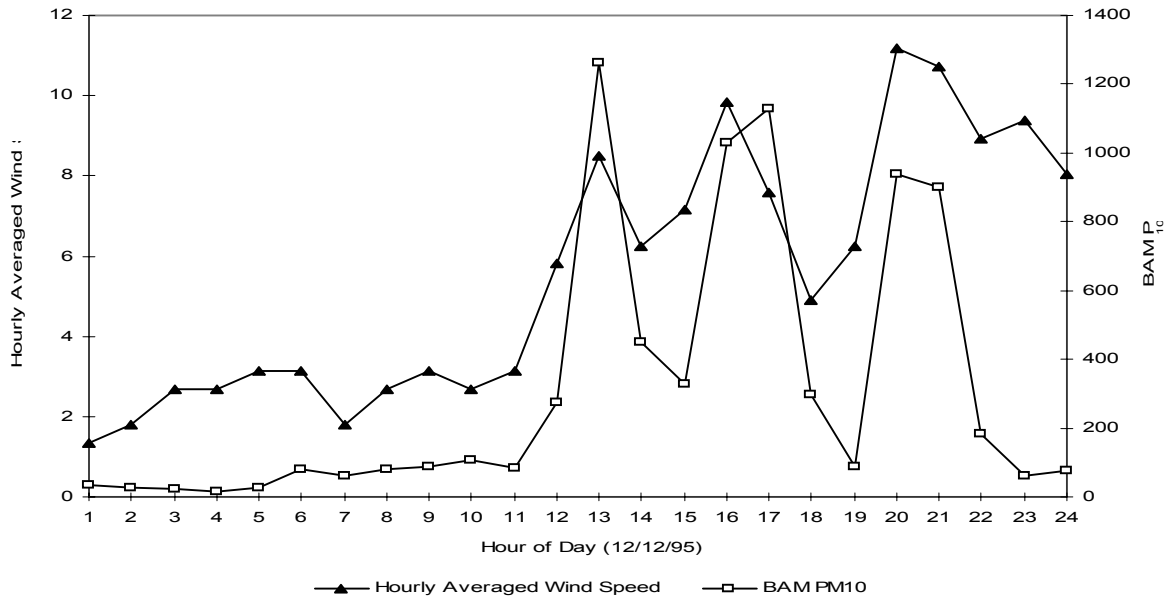


Figure 7-2. Co-variation of hourly PM_{10} concentrations ($\mu g/m^3$) measured by a Beta Attenuation Monitor (BAM) and hourly wind speed at a location in northern Las Vegas downwind of major construction sites on 12/12/95 (Chow and Watson, 1997).



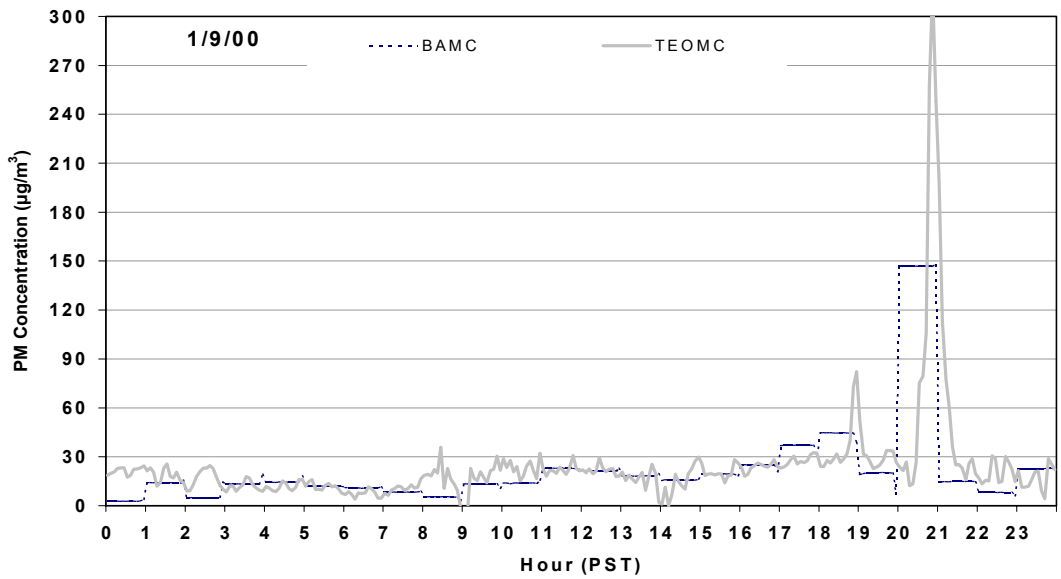
Fugitive dust contributions can occur over short durations when wind speeds are low as illustrated by the coarse aerosol concentration shown in Figure 7-3 for the Fresno "Supersite" during winter when wind speeds were less than 2 m/s. The coarse aerosol concentration is calculated as the difference between the PM_{10} and $PM_{2.5}$ concentrations measured with either a beta attenuation monitor (BAM) or a Tapered Element Oscillating Microbalance (TEOM). The TEOM is capable of 5-minute averages while the BAM temporal resolution is no less than an hour. The 24-hour average coarse particle concentrations were 22.1 $\mu g/m^3$ for the BAM and 23.9 $\mu g/m^3$ for the TEOM, with 28% of the BAM concentration contributed during the one-hour period between 2000 and 2100 PST. 23% of the TEOM coarse concentration was contributed by the large peak that slightly lags the BAM peak during this period. These results clearly indicate that there was an obvious fugitive dust episode at night. It was not wind blown and there was no apparent cause, such as a pile of dirt or loose dust on the pavement, that was visibly apparent on the following day. While this event would have caused visibility degradation, it occurred over a short time period and at night, and so was not observed. Nevertheless, a 24-hour sample would show a large contribution from this high dust event that would be integrated over the entire sampling period rather than treated as the specific, but unknown, event that it was. An initial examination of the Fresno "Supersite" data shows that occurrences such as these occur for a small fraction of the samples. Undocumented emissions take place that may substantially affect the interpretation of ambient concentrations and their contributions to visibility and health effects.

7.2 Limitations

Individual emissions events that affect daily or hourly dust contributions cannot be adequately documented in any practical inventory. They can be detected, however, by fast time-response

particle measurements, even though this detection may not contain sufficient resolution to identify the location and nature of the actual event.

Figure 7-3. Coarse aerosol concentration measurements from the Fresno "Supersite" on 1/9/00 (Watson et. al., 2000).



7.3 Information Gaps

Changes in activities and surfaces surrounding Class I areas have not been adequately documented or deduced from existing data bases.

Meteorological information such as wind speed, temperature, and relative humidity that might indicate surface conditions, vegetative cover, and erosion potential have not been used to resolve emissions in a multi-state region around Class I areas on monthly or seasonal time scales.

7.4 Recommendations

Identify and evaluate agricultural, meteorological, and land use data bases for selected regions around Class I areas to determine the extent to which the California methodology (Cambell and Shimp, 1998), that looked at seasonal variations at the county level scale, can be adapted to a larger scale.

Analyze existing nephelometer data from the IMPROVE network in conjunction with elevated soil contributions to identify potential events that might have occurred during the 24-hour sampling period. Adapt nephelometer sampling schedules and wind measurements to 5-minute averages to better detect short emission pulses that might arise from fugitive dust and other sources.

Operate continuous particle monitors, preferably ones that are sensitive to the coarse particle fraction, at 30-minute or shorter time resolution at selected IMPROVE sites. Examine these data

to determine the fraction of a 24-hour sample that is contributed by short duration events and which are more consistent throughout the daytime viewing period where haze is of concern to visitors. Use these events to develop a statistical distribution of occurrences of unknown events and evaluate the magnitudes of these events relative to longer-term emissions estimates.

8. SPATIAL ALLOCATION OF FUGITIVE DUST EMISSIONS ARE IMPORTANT (FINDING #7)

Better spatial surrogates than are currently in use are available for estimating fugitive dust reservoirs, locations, dust-generating activities, and temporal changes in surface properties and surroundings. Use of these spatial surrogates would provide better emission estimates.

8.1 Technical Support of Finding #7

Most inventories use national activity surrogates that are remotely related to reservoirs and activities that create fugitive dust emissions. Meteorological surrogates that affect emissions (moisture, wind speed) are averaged temporally and spatially. Reservoirs are assumed to be unlimited, rather than depletable. Improvements can be made in all of these areas by using available, or soon to be available, spatial surrogates. These include road maps, vegetative land cover maps, land use maps, soil maps, and spatially and temporally resolved meteorological data.

8.1.1 Spatial Surrogates for Suspendable Particle Reservoirs

These spatial surrogates must be related to the availability of dust for suspension. This is currently done by estimating the "silt" fraction or loading in surface dust. 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 which is ~2.65 g/cm^3 for most minerals (Hinds, 1999). 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 US and are being made available in the STATSGO and SSURGO data bases. The STATSGO data base provides county-level information while the SSURGO data base provides finer spatial resolution and a larger number of measured parameters for several of these smaller surface areas. There are spatial gaps in these data bases, but these are being filled. One may be able to extrapolate soil characteristics from nearby regions to those areas that are not yet surveyed. The particle sizing procedure most commonly followed for soil surveys creates a soil/water suspension in which soil aggregates are broken into their component parts prior to sieving (American Society for Testing and Materials, 1997, 1998). 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. This differs from the method used by EPA AP-42 (Appendix C-1) that uses a hard sieving, dry sieving method to define silt content. This difference in definitions has precluded the use of soil survey data bases for estimating silt content.

Carvacho et. al. (1996) and Ashbaugh et. al. (2000) have developed a five step approach that directly relates the PM_{10} reservoir in a surface loading to parameters that are included in the STATSGO and SSURGO soil surveys. A pilot study in California demonstrates the methodology and illustrates that it has great potential to estimate PM_{10} reservoirs from soil surveys. The first step involved studying soil survey and land use maps to determine where and when soils with different properties and activities with a potential for emissions occur. Figure 8-1 shows that the 50 samples acquired in the pilot study represented a variety of different soil types typical of the alluvial deposits in central California. A much wider range of compositions is expected for areas around Class I areas, especially in the desert southwest. The second step involved examining activities that might create reservoirs or suspend dust. In the pilot study, it was found that reservoirs are created during September through November when crops are harvested and that windblown erosion is most prevalent in April after land surfaces have dried but wind speeds during frontal passages are high.

The third step involved collecting samples of the surface loadings. These samples were taken to a depth that represents the available reservoir containing suspendable PM_{10}. The exact depth for a given area needs to be better defined, as the effective reservoir changes with suspension depending on surface roughness, friction velocity, availability of large clods to shield the suspension of smaller particles, and the availability of 84 μm particles (i.e., the median saltation diameter) that cause reservoir creation via saltation. Better guidance is needed on how to characterize these parameters and to translate that characterization into an effective depth of the soil surface from which particles might be suspended. In the fourth step, the samples were suspended in a laboratory chamber and sampled through size-selective inlets onto filters which were subsequently weighed. Carvacho et. al. (1996) utilized a constant air flow through a fluidized bed of a specified amount of the soil sample. This air stream was sampled using several filters in parallel, with each subsequent filter having a longer sampling duration than the previous one. These filters were weighed and the results are shown in Figure 8-2. This procedure might be made more efficient for a large number of samples by use of continuous PM monitors, such as a beta attenuation monitor (BAM) or a Tapered Element Oscillating Microbalance (TEOM). Figure 8-2 shows that the PM_{10} availability levels off after a few minutes of sampling, and this maximum value is termed the PM_{10} index. Note that for a given dry silt loading (i.e., that used in AP-42 emission factors), there is more than a factor of 4 variability in the PM_{10} potential for the soil samples depicted in Figure 8-2.

The final step involved deriving relationships between soil survey variables and the "PM_{10} index" defined by the maximum value achieved in Figure 8-2 for the samples taken, then spatially allocating the findings using soil maps of the area. Figure 8-3 shows that, for the samples used in the pilot study, there is a good relationship between the PM_{10} index and all of the soil texture variables reported in soil surveys. Figure 8-4 shows a reasonable relationship between the PM_{10} index and the AP-42 dry silt measurement, but there is much more scatter than the relationships to the wet-sieve quantities depicted in Figure 8-3.

Figure 8-1. Soil composition triangle, commonly used by geologists to classify soils, showing the composition of fifty samples taken from different parts of central California (Ashbaugh et. al., 2000). These samples represent most of the compositions reported in soil surveys for the alluvial deposits in California's San Joaquin Valley.

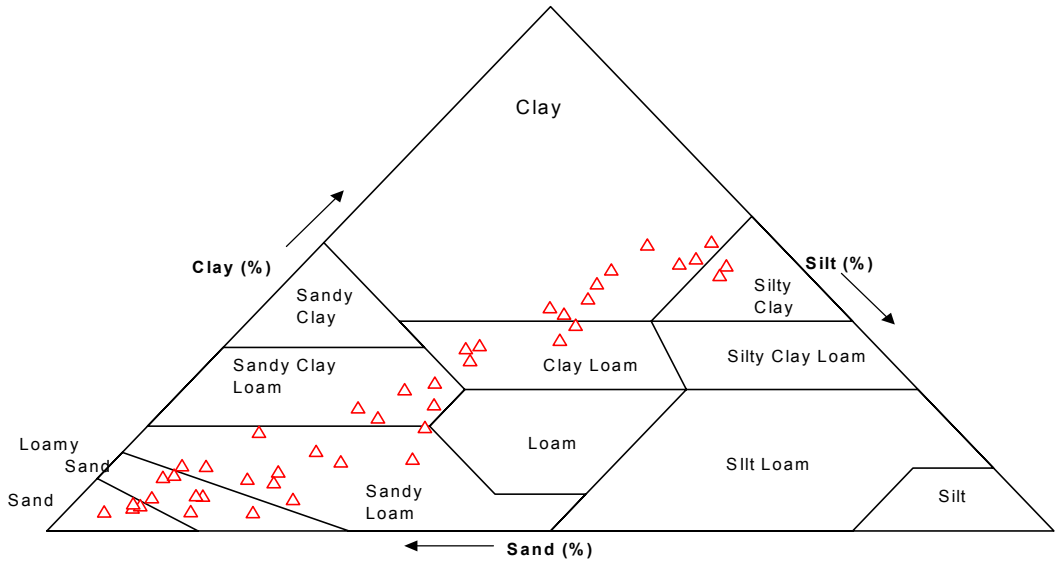


Figure 8-2. Weight of PM_{10} generated per gram of AP-42 dry silt content for different soil samples from central California (Carvacho et. al., 1996).

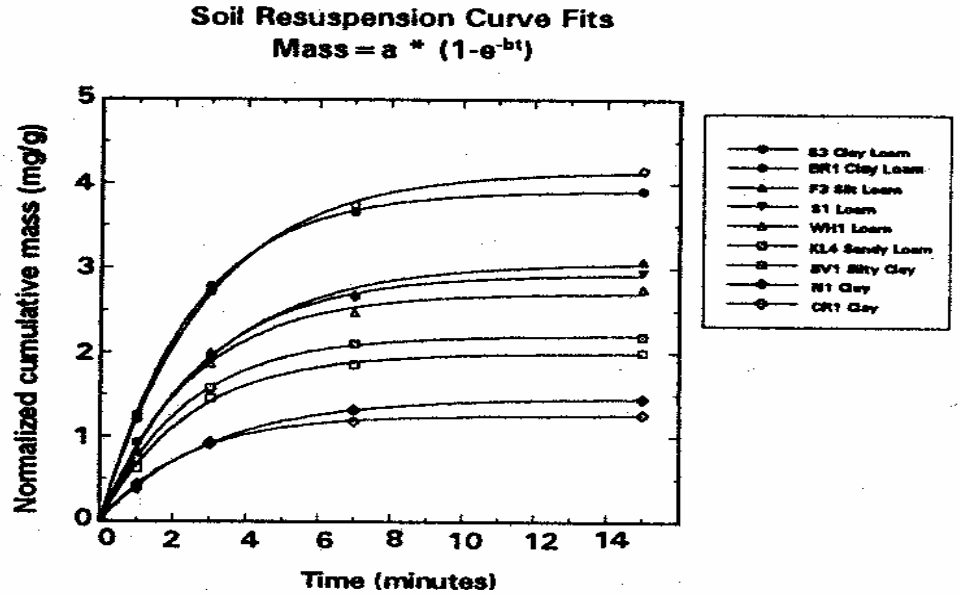


Figure 8-3. PM_{10} index vs. silt, clay, and sand parameters reported in soil surveys (Ashbaugh et. al., 2000). Results are shown for sieved fractions less than 75 µm and less than 2000 µm.

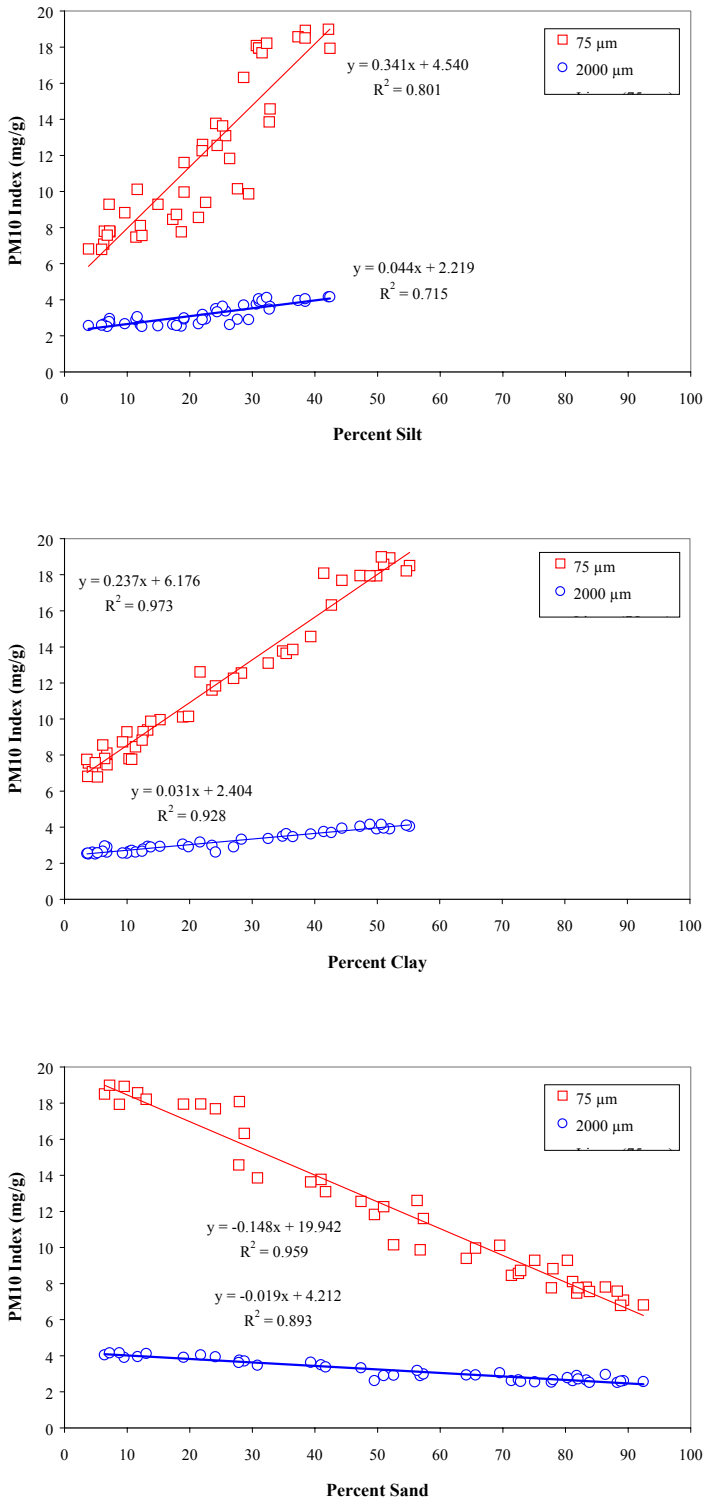
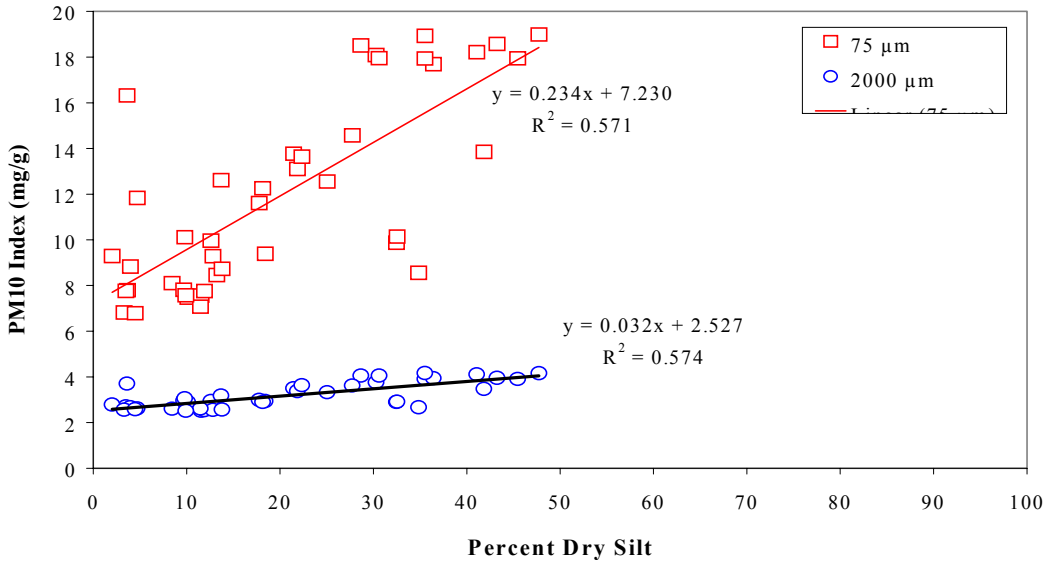


Figure 8-4. Relationship between PM_{10} index and AP-42 silt content (Ashbaugh et. al., 2000). Results are shown for sieved fractions less than 75 µm and less than 2000 µm.



While this method applies to resuspension of dust due to wind erosion of open spaces, it is not directly applicable to roadway reservoirs, the other major source of fugitive dust. AP-42 measurements of silt loading and silt content involve periodic sampling from representative roads which are then used to calculate emissions. These silt loadings have been shown to be highly variable in time and space, and the labor required for their acquisition mitigates against frequent sampling that covers a wide spatial extent. Kuhns et. al. (2001) have developed a method named the TRAKER method that relates real-time measurements to silt loading using one continuous particle measurement device located in the wheel well of a vehicle and a second unit located on its hood. Forward-scattering nephelometers were used by Kuhns et. al., 2001, but other instruments more specific to different particle sizes could be applied. Since suspendable dust remains close to the ground, the difference in readings between the two nephelometers is proportional to the amount of suspendable dust on the roadway surface. Figure 8-5 shows the variability determined by this method in a pilot study in Las Vegas, NV. The TRAKER was calibrated against AP-42 silt loadings determined for samples taken in the study area for a variety of visible surface loadings and traffic types and volumes. Figure 8-6 shows a reasonable relationship between the continuous TRAKER measurements and actual silt loadings. This method could be applied to roadways in and around Class I areas to rapidly and efficiently determine statistical distributions of surface loadings and to identify hot spots. Patrol and maintenance vehicles could be equipped with such devices so that data could be acquired during the normal rounds of park personnel. An ancillary benefit would be to identify hot spots, possibly caused by road deterioration or trackout, to which emissions reduction measures might be applied.

Figure 8-5. Variability in silt loading in Las Vegas using the Testing Re-entrained Aerosol Kinetic Emissions from Roads (TRAKER) method (Kuhns et. al., 2001).

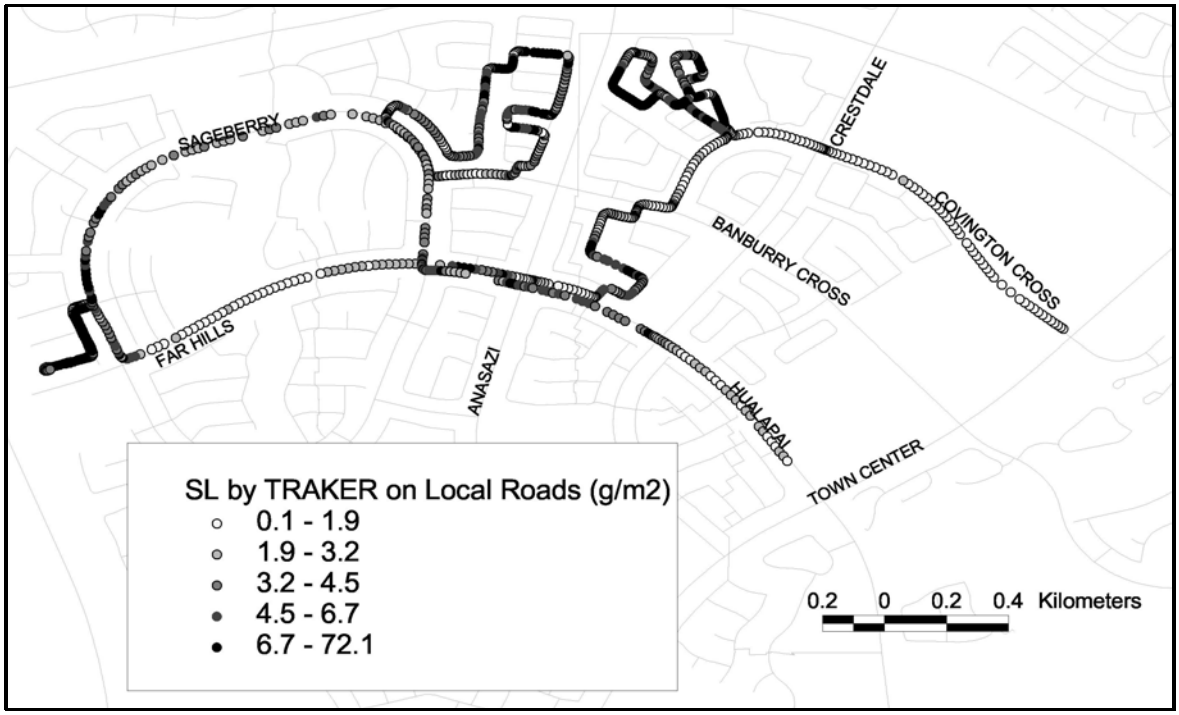
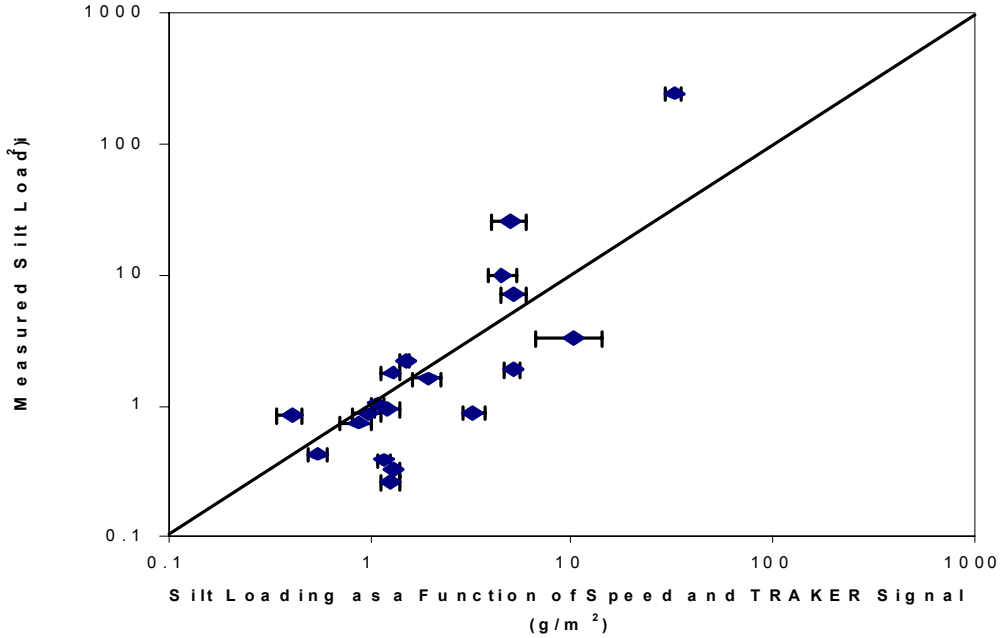


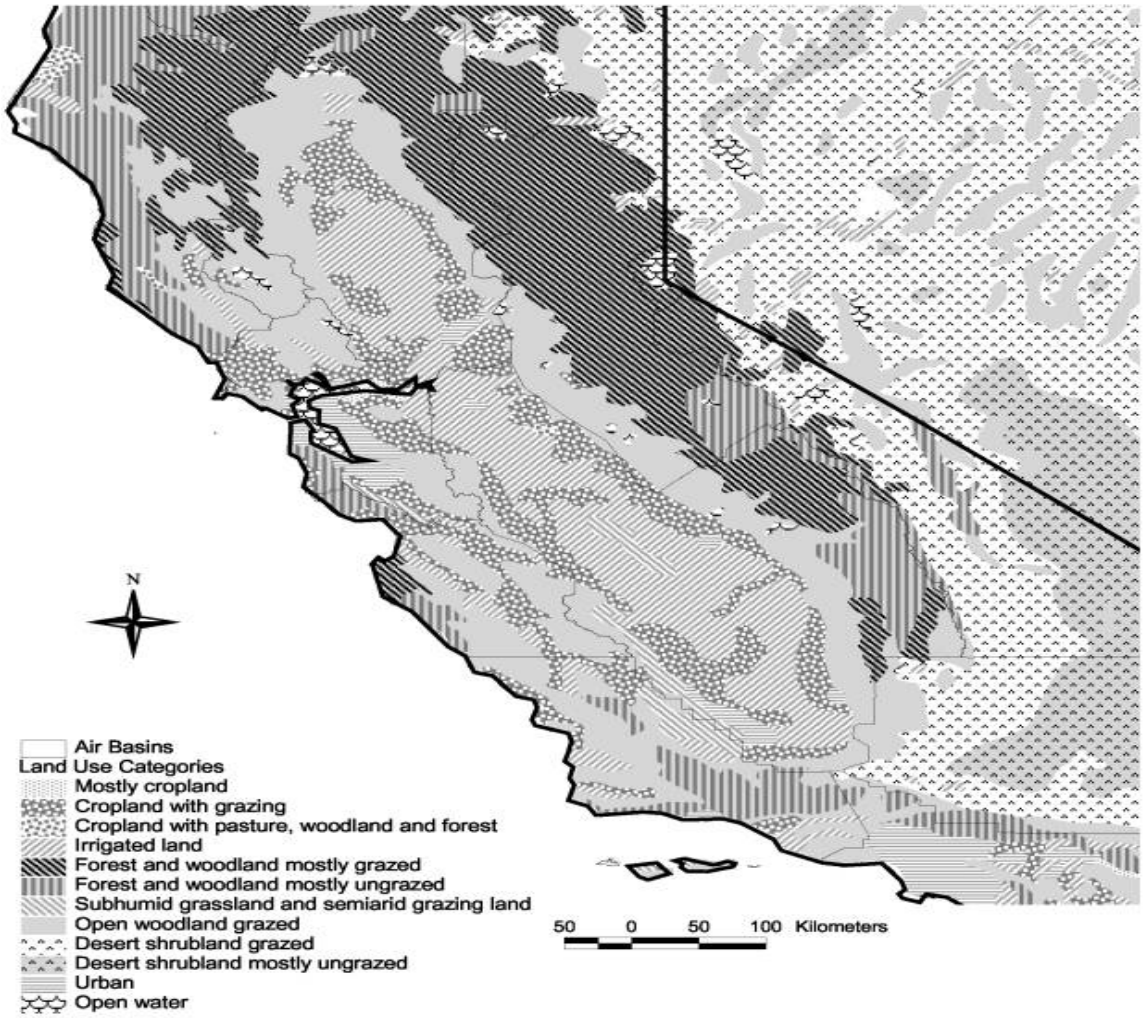
Figure 8-6. Relationship between silt loading measured by the AP-42 method and speed-adjusted silt loading by TRAKER (Kuhns et. al., 2001).



8.1.2 Spatial Surrogates for Activities that Create, Enhance, or Reduce Reservoirs

Once the PM_{10} suspension potential for a soil surface is known, it is necessary to determine what happens on that surface to generate or deplete reservoirs. The most important spatial surrogate is land use. Bare land has more potential to create dust than surfaces with vegetation. Land use (specifically terrain) affects winds near the surface and indicates obstructions that increase particle deposition near the point of emissions. Figure 8-7 shows an example of the USGS land use maps that are available in digital form and could be used as a starting point. More of this information is becoming available through satellite surveys that show how surface covers change with time. Roadway maps, especially those for unpaved roads, need to be superimposed on soil use maps because these are the most likely surfaces to create and maintain dust reservoirs. Inexpensive mapping software (e.g. DeLorme) products locate these roads and identify them as unpaved. The extent to which these base data can be superimposed on soil property maps and related to traffic volumes needs to be investigated.

Figure 8-7. Example of USGS land use maps for California.



8.1.3 Spatial Surrogates for Activities that Suspend Available Dust

The major dust suspending activities that affect non-urban emissions are excessive winds, traffic, and agricultural plowing. Winds aloft are estimated by continental-scale models by the National Weather Service (NWS), but these do not adequately represent wind speeds near the surface where dust is suspended. Regional climate centers have compiled and integrated wind data from many different meteorological networks that could be used in diagnostic models to estimate wind speeds near the surface. These data bases include NWS measurements, most of which are near airports and not relevant to exposed surfaces. They also contain statewide agricultural meteorological measurements that are relevant. The AZMET data base in Arizona and the CIMIS data base in California measure winds one to two meters above the surface for the express purpose of estimating evapotranspiration and wind erosion. Many other states have similar data bases. The RAWS (Remote Automated Weather Station) fire weather monitors are located throughout the arid west, especially in those locations that might experience regional erosion. Transportation demand models have been developed for interstate highways in many states that can be used for traffic estimates. Other surveys for less traveled roads have been made, or could be made, to determine what actually happens on unpaved roads. Crop calendars are well established in different agricultural communities, but these vary by climate, crop, and state. These need to be surveyed and compiled.

8.2 Limitations

There is an institutional reluctance to use available spatial surrogates for reservoirs and activities. This reluctance is somewhat justified based on the lack of spatial coverage or lack of resources. The extent of these deficiencies has not been defined. It might be found that the value of the more accurate information justifies the cost of using and judiciously extrapolating existing data. Decision-makers cannot make this judgment until they know the availability of spatial surrogate data bases, the cost of obtaining and using them, and the potential improvements in emission estimates that might result from that use.

8.3 Information Gaps

Available data bases have not been adequately evaluated to determine their spatial coverage, availability, and applicability to determining reservoirs and activities over large regions surrounding Class I areas.

Several areas do not yet have soil surveys, and the extent to which existing soil surveys can be applied to the unknown areas has not been assessed.

8.4 Recommendations

Catalog, describe, and evaluate spatial data bases such as soil surveys, digital road maps, satellite and other land use data, meteorological measurements and models to interpolate and extrapolate measurements and traffic demand estimates; and determine the availability and costs of these data and identify technical and cost impediments for using them to improve fugitive dust inventories.

Develop and apply a systematic program to sample representative soils; determine their PM_{10} and $PM_{2.5}$ indices; relate these to soil texture properties in soil surveys; and use these to estimate

suspendable particle reservoirs on open lands. This could be done in conjunction with the source profile studies recommended for Finding 11 (see Section 12).

Develop and apply a practical method to obtain continuous roadway dust loadings; evaluate the potential to apply this methodology during normal driving cycles of park personnel and others; and evaluate these data to determine statistical distributions of surface loadings and determine how they change with different variables.

Develop and apply a flexible GIS emissions modeling structure that continuously acquires and updates spatial surrogates from existing and planned data bases and propagates this new information into better estimates of reservoirs, dust-suspending activities, and attenuation from obstacles near the point of emissions.

9. THE FINE FRACTION OF FUGITIVE DUST EMISSIONS IS NOT ADEQUATELY CHARACTERIZED (FINDING #8)

Few empirical measurements exist for characterizing the $PM_{2.5}$ fraction of fugitive dust emissions (i.e., that fraction that has the longest residence time in the atmosphere and that has the largest impact on visibility degradation). The $PM_{2.5}$ fraction often behaves differently than the coarse fraction with respect to dispersion and deposition.

9.1 Technical Support of Finding #8

Most emission factor studies (especially for paved and unpaved roads) measured total suspended particulates (TSP) since that was the regulatory standard at the time the measurements were made. These measurements were augmented with a modest amount of PM_{10} data when PM_{10} became the regulatory standard coupled with some measurements of $PM_{2.5}$. However, few direct measurements of fine particles exist for wind erosion. When fine particles were measured, older measurement devices with imprecise cutpoints or that did not adequately account for particle bounce within the device were used. Any particle bounce would have caused the $PM_{2.5}/PM_{10}$ ratio to be too high. EPA has recently reviewed the collected information on $PM_{2.5}/PM_{10}$ ratios developed during these measurement programs to determine the effects particle bounce may have had. That review resulted in modifications to the $PM_{2.5}/PM_{10}$ ratio used to estimate emissions for the NET inventory. When particle size distributions were measured, they were only collected at only one height in the profiler array. As a consequence, the change in the vertical distribution of particle sizes (or particle size ratios) cannot be adequately determined.

9.2 Limitations

Significant uncertainties exist with respect to the vertical distribution of fine particles from fugitive sources. Only limited experiments have been performed that measured the vertical profile of emissions for fine particles. Measurements in the past have typically been at only one or two heights above the ground. In addition, where these measurements have been made, the samplers used have sometimes had particle bounce problems leading to overestimates of the fine particle fraction.

9.3 Information Gaps

Insufficient information exists on directly measured emissions of fine particles. The vertical distribution of fine particles is also poorly understood.

9.4 Recommendations

Research efforts to better characterize the fine particle fraction are clearly needed. In order to effectively quantify the vertical flux of fine particle materials into the atmosphere, information on the vertical particle size distribution is needed. In addition, direct determination of emission factors for fine particulate matter (as opposed to values inferred by single instrument measurements during exposure profiling designed to evaluate TSP or PM_{10} emission factors) is needed to effectively characterize fine particulate emissions from fugitive sources.

10. AIR QUALITY MODELS NEED TO INTEGRATE METEOROLOGY AND THE EMISSIONS PROCESSES (FINDING #9)

Regional scale air quality models need to integrate all the appropriate meteorological processes with the fugitive dust emissions generation processes. Meteorological parameters used for estimating transport and dispersion should be combined with the fugitive dust emission estimates for both mechanically-generated and wind erosion-generated conditions. For example, to model windblown dust, if time-averaged wind speeds are used, it is important to account for smaller time-scale wind gusts. The use of air quality dispersion models that account for injection heights and deposition losses should allow one to distinguish the relative impact from near field (i.e., local) emissions versus far field (i.e., regional) emissions on ambient air quality.

10.1 Technical Support of Finding #9

For the most part, dispersion modeling of fugitive dust has been done as part of other modeling analyses. Dispersion models include the relevant chemistry and physics of the atmosphere describing transport, turbulent diffusion, chemical reaction, emissions, and wet and dry deposition. The relevant atmospheric diffusion equation (also called the species continuity or advection/diffusion equation) is shown below:

$$\frac{\partial c_i}{\partial t}+\frac{\partial (uc_i)}{\partial t}+\frac{\partial (vc_i)}{\partial t}+\frac{\partial (wc_i)}{\partial t}=\frac{\partial}{\partial x}\left(K_x\frac{\partial c_i}{\partial x}\right)+\frac{\partial}{\partial y}\left(K_y\frac{\partial c_i}{\partial y}\right)+\frac{\partial}{\partial z}\left(K_z\frac{\partial c_i}{\partial z}\right)+R_i+S_i-D_i-W_i \qquad (10\text{-}1)$$

The first term above describes the time dependence of the concentration of species i. The next three terms are the transport (advection) resulting from a wind with components u, v, and w. On the right side of the equation the first three terms represent the turbulent diffusion. R represents the rate at which chemical reactions produce species i, S is the emission rate, D is dry deposition, and W is wet deposition. For fugitive dust, the R term above can be eliminated since fugitive dust is a directly emitted inert species. The diffusion equation can be solved for any combination of inputs using a fine-difference solution approach to the partial differential equation. These so-called grid-based Eulerian models are now relatively common in air quality work (USEPA, 1999). Examples of grid models are the Urban Airshed Model (UAM), REMSAD (Systems Applications International, 1998), and CMAQ (USEPA, 1999). These models are called grid models because the area of interest (modeling domain) is divided up into a three-dimensional grid of cells each representing a specific portion of the atmosphere. For small areas, these grid cells can be very small. However, for large areas which are regional in scope, grid cells can be quite large (e.g., 36 km x 36 km for REMSAD or CMAQ in a country-wide application). One of the problems with grid models is that they instantaneously mix emissions throughout a grid cell. For fugitive dust area sources such instantaneous numerical diffusion may completely miss the significance of an air quality effect. For example, if a grid cell is 36 km x 36 km, emissions would be mixed uniformly throughout this cell, even though in reality the dust plume would be limited to a much smaller area.

For a simple situation such as the case of a constant wind, one can solve the above advection/diffusion equation utilizing a Gaussian formulation. Sometimes a Gaussian approach is embedded within a grid model to account for the sub-grid diffusion mentioned above. This is called a plume-in-grid treatment (USEPA, 1999). A Gaussian treatment can be combined with a

trajectory analysis to determine the incremental impact of regional sources while accounting for spatially and temporally changing meteorology (National Research Council, 1993). CALPUFF is an example of such a model.

One of the shortcomings of most modeling exercises is that fugitive dust emissions estimates are typically only provided as annual or seasonal averages. However, the highest fugitive dust emissions generally occur during short-term meteorological conditions that cause the greatest vertical turbulence and hence enhanced dilution. Thus, fugitive dust is a very different type of emission source compared to a power plant with a relatively fixed load or an urban area mobile source with its twice-daily rush-hour peaks and weekday/weekend differences. Fugitive dust emissions are strongly affected by wind speed, humidity, precipitation amount (both before and during the period of emission), time since previous precipitation event, and temperature. Fugitive dust emission rates are expected to be highly variable, with major peaks occurring during dry, strong wind events. Since sophisticated air quality models track all of the above meteorological parameters, it is feasible to couple/integrate these meteorological parameters with the fugitive dust emission estimates. Furthermore, it is important that these air quality models account for transport and dispersion of the fugitive dust. Existing air quality models generally do not simulate violent wind events such as dust devils and thunderstorms which can provide sufficient energy to disturb natural surfaces and to entrain dust from already disturbed surfaces. However, by changing the temporal and spatial scale (e.g. increasing the spatial resolution from, say, 36 km to 1 km) of a meteorological model such as MM-5 that is currently commonly used to calculate meteorological input fields for air quality models, it may be possible to model such events on a crude basis. This interaction between meteorology and emissions, assuming it accounts for injection heights, deposition losses, and horizontal impaction losses, should allow one to distinguish the relative impact from near field (i.e., local) emissions versus far field (i.e., regional) emissions on ambient air quality (i.e., aerosol concentrations and visibility degradation).

The rate of loss of fugitive dust during dry conditions can be calculated from a simplified version of the dispersion equation. The rate of fugitive dust loss is:

$$\frac{c}{c_0} = e^{-kt} \tag{10.2}$$

where k is the ratio of the dry deposition velocity (v_d) to a characteristic height: $k = v_d/h$. One can calculate the rate of dry deposition for the scenario of a dust plume released at ground level during neutral stability (D stability) conditions. If the wind is light (5 km/h), it takes one hour to transport the dust 5 km. Vertical diffusion (σ_z) is 30 m at 1 km and 80 m at 5 km (Turner, 1970). The plume height is twice σ_z, so the average plume height over the 5 km of transport is about 110 m. The deposition velocity of coarse and fine particulate matter can be determined from the deposition velocity diagram in Seinfeld and Pandis (1998).

10.2 Limitations

A serious limitation of traditional Gaussian models is their assumption of steady state emissions. Since fugitive dust emissions are highly variable in nature, traditional Gaussian models need to be revised to handle intermittent emissions. If all of the relevant inputs to the diffusion equation are quantified within a specified accuracy, the diffusion equation can provide realistic estimates

of the concentrations throughout an area or a region. The problems occur when important inputs (listed below) are uncertain.

Advection For example, consider the three terms to the left of the equals sign in the diffusion equation. If the u, v, w components of the wind are not known accurately in every grid cell for each time of the simulation, the model output is less likely to be realistic. In some situations, winds are fairly uniform and steady. However, in other situations winds are variable and can change with height and location in the region.

Diffusion Atmospheric turbulence is one of the key ways that species are initially dispersed. Turbulence varies strongly depending on the time of day, the amount of insolation, and wind speed, and surface roughness. Quantification of turbulence is uncertain.

Emissions Since chemical reactions are not germane to fugitive dust, the next important input term in the diffusion equation is emissions (S). These must be quantified for each time and location in the model run (x,y,0,t). Ideally one would also want information regarding the initial dilution of material. For example, vehicles traveling down paved or unpaved roads generate emissions with considerable mechanical turbulence, often tens of meters high, immediately after release. Currently dispersion models are not inter-linked with emissions models. It would be beneficial to have an integrated model where the meteorological conditions drive both the emissions and the diffusion calculations. Note: fugitive dust emissions would be higher during windy and dry periods.

Deposition Dry and wet deposition are effective scavengers of fugitive dust. However, the time period for removal can be very long. During periods of precipitation, fugitive dust is generally neither emitted nor capable of remaining in the atmosphere.

10.3 Information Gaps

Dispersion models may be effective as a diagnostic tool in studying the validity of fugitive dust emission estimates. However, there are a number of uncertainties in dispersion modeling, generally pertaining to the fact that one simply cannot determine the exact winds, turbulence, emissions, and deposition rates that pertain to a given situation. It is acknowledged that dispersion models should not be used as the "gold standard" in any scientific work related to fugitive dust. However, when used judiciously, dispersion models are useful tools to diagnose potential problems with the inputs (such as the fugitive dust emission estimates) used in the model. For example, if a model does well for one species and not for another, this may imply that the emission inputs are incorrect for that species and/or deposition of one species is markedly different from the other species.

Recently EPA's OAQPS has recommended reducing previous estimates of fugitive dust emissions by a factor of two to four. This decision is based on modeling results obtained by Latimer (2000) for Denver and the Grand Canyon using several different air quality models including the Climatological Regional Dispersion Model (CRDM). Latimer's results indicated that the models performed well for gas phase species whereas the fugitive PM_{10} dust concentrations were significantly overestimated. This may be due to the fact that the models did

not take into account aerosol loss mechanisms (i.e., the difference between horizontal and vertical flux emission estimates).

10.4 Recommendations

Reconcile model predictions with measurements by incorporating an interim method for accounting for near source removal of particles into regional models. For those cases where the model predicted ambient concentrations that were considerably larger than the observed downwind concentrations, this will involve running the model utilizing estimates of the vertical fugitive dust emissions flux rather than the local-scale horizontal flux as inputs for the model.

Incorporate more refined particle removal estimation methods into regional models as they become available.

11. DISTURBED SURFACES PRODUCE SIGNIFICANTLY MORE FUGITIVE DUST THAN UNDISTURBED SURFACES (FINDING #10)

In general, undisturbed surfaces produce much less dust than disturbed surfaces. This is because the undisturbed surface usually requires considerably higher wind speeds to become a significant emission source. A surface having a lower threshold velocity produces much more dust than a surface having a higher threshold velocity. The primary influence of disturbing a surface is to lower the threshold velocity and increase dust emissions.

11.1 Technical Support of Finding #10

A simplified model for wind erosion saltation flux is given by Equation 11-1:

$$E = k' C_D^{1.5} \frac{\Delta T}{L} \int_{u_t}^{\infty} U^3 f(U) dU \tag{11-1}$$

where E is the expected saltation flux (particles moving in hopping motions), k' is a constant, C_D is the drag coefficient, ΔT is the time interval considered, L is the length of the source area measured parallel to the wind direction, u_t is the threshold velocity (with erosion increasing as the cube of wind speed U), and f(U) is a probability density function of the wind speed. The threshold velocity is that wind velocity at which the erosion process starts. For wind speeds lower than the threshold value, there is no erosion. This threshold value occurs when the aerodynamic lift and drag equal the forces of gravity and cohesion holding the particles in position. Because the wind erosion proceeds as the cube of wind speed after the threshold velocity is exceeded, the probability distribution of wind speed is very important. This model assumes an unlimited supply source. That is, the disturbance is assumed to be large enough that the particle emissions are described by Equation 11-1. To illustrate this point, using a simple but plausible probability density function, f(U) is taken to be the Rayleigh distribution given in Equation 11-2:

$$f(U) = \frac{\pi U}{2\overline{U}^2} \exp\left[\frac{-\pi U^2}{4\overline{U}^2}\right] \tag{11-2}$$

where $\overline{U}$ is the mean wind speed. The Rayleigh distribution successfully simulates empirical probability density functions of wind speed in many areas having wind erosion problems (Cowherd , et. al., 1984). A solution for Equation 11-1 is given as:

$$E = k' C_D^{1.5} \Delta T \left[\frac{4}{\pi}\right]^{1.5} \overline{U^3} [1 - \Gamma(2.5, z)] \tag{11-3}$$

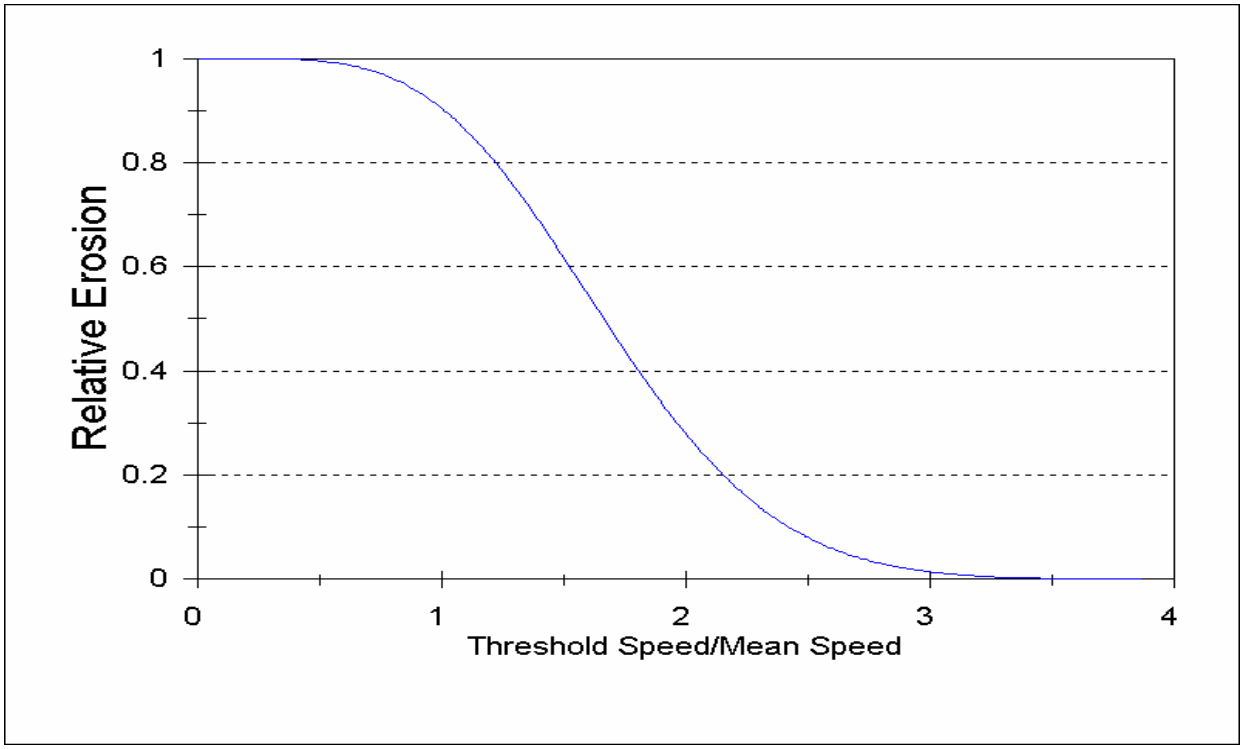
where $\Gamma(2.5,z)$ is an incomplete gamma function and:

$$z = \frac{\pi}{4} \left[\frac{u_t}{\overline{U}}\right]^2 \tag{11-4}$$

For a given location, the mean wind speed is constant so that the variable part of the right side of Equation 11-3 is proportional to the complement to the incomplete gamma function. Details on the incomplete gamma function are given by Abramowitz and Stegen (1964). The expected saltation flux (i.e., relative erosion calculated from Equation 11-1) versus the ratio of threshold

wind speed to mean wind speed is plotted in Figure 11-1. Figure 11-1 shows that the relative erosion changes very little for ratios of threshold wind speed to mean wind speed less than 0.5 and is close to one. However, the relative erosion decreases rapidly for a ratio greater than one.

Figure 11-1. Relative Wind Erosion vs. Ratio of Threshold Wind Speed to Mean Wind Speed



The influence of the probability density function of wind speed and threshold velocity for saltation flux on the intensity of wind erosion is enormous. For the Rayleigh probability density function, Equation 11-3 shows that the saltation flux may be thought of as a potential dust emission depending on mean wind speed cubed times a power of the drag coefficient and times a function of mean wind speed and threshold wind speed. This function (given in Equations 11-3 and 11-4) is very sensitive to the threshold wind speed. Since disturbance lowers the threshold velocity for wind erosion and the total emission of dust is assumed to be proportional to E (the total horizontal saltation flux) any disturbance can greatly change the total production of dust.

For a given location, the probability density function of wind speed, k', and L do not change with disturbance for a given source area. However, the threshold velocity u_t and drag coefficient C_D would be expected to change. A change which significantly decreases the drag coefficient would be expected to decrease the potential dust emission, while increasing the drag coefficient would be expected to increase the potential dust emission. Gillette et. al. (1980) found that the disturbance of vehicular traffic on natural surfaces decreased the threshold velocity. At the same time, the disturbances usually acted to smooth the surfaces which decreased the drag

coefficients. For almost all of these surfaces, however, the large decreases of threshold velocity were sufficient to greatly increase dust emissions even though drag coefficients were decreased.

11.2 Limitations

The limitations of the above analysis include the need to accept the following assumptions:

- The horizontal flux of sand is roughly proportional to the cube of the wind velocity. In fact, the relationship of wind velocity to flux of sand moved in a wind erosion event is more complicated. However, for the qualitative relationship developed here, it is approximately true.
- The probability density function (pdf) of wind speed is often described as a Rayleigh pdf. For those locations for which the pdf is not adequately described by a Rayleigh distribution, the results will be in error. However, for this qualitative demonstration, the Rayleigh distribution is a good approximation for commonly found pdf's.
- For the case of a small disturbance, the dust emissions will be supply-limited and the estimate of E will be too large. That is, if a disturbance liberated a small amount of particulate material from a crust, only this small amount of material could become airborne.
- The assumption of a linear relationship between vertical flux of PM_{10} and horizontal flux of sand-sized particles has been shown to be highly dependent on the texture of the surface material. The linearity is somewhat controversial, although most data at this time support it.

11.3 Information Gaps

A major information gap for the above analysis is whether the disturbance will produce a supply-limited source or a supply-unlimited source of dust. In addition, very little is known about how different kinds of disturbance affect the aerodynamic roughness height.

11.4 Recommendations

Although a fairly large information gap exists, the conclusion that the disturbance of the surface increases the emissions of dust is quite robust. An effort that would help to improve predictions of the effect of disturbance on dust emissions would be the quantification of supply of loose dust material available to be eroded as generated by the disturbance. Specific recommendations include: cataloging existing studies and conducting studies to determine if different surfaces are supply-limited or supply-unlimited (i.e., unlimited particle reservoir); and conducting studies to determine the effect that different kinds of disturbance have on the aerodynamic roughness height of different surfaces.

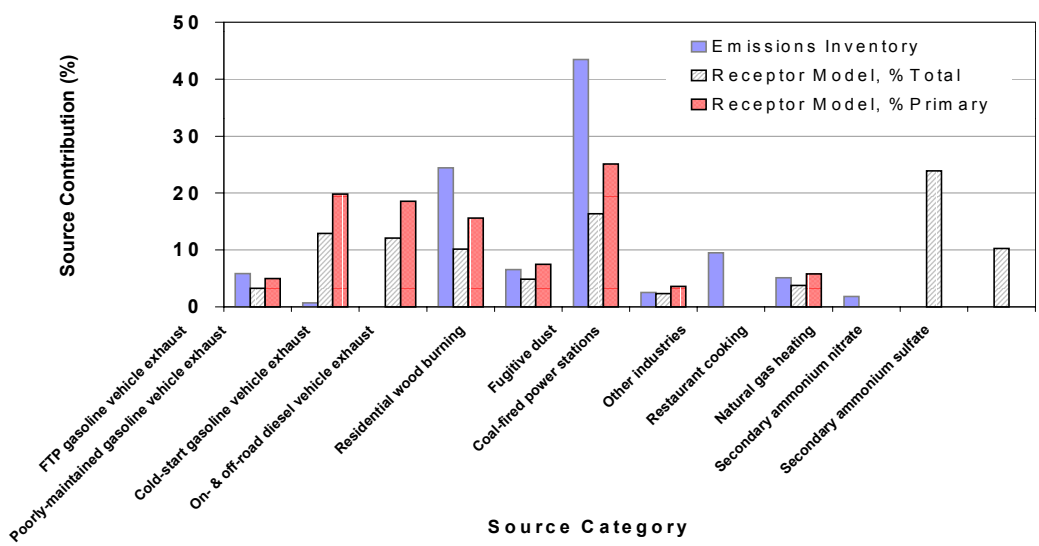
12. RECEPTOR MODELS MAY BE USED TO DISTINGUISH CONTRIBUTIONS FROM DIFFERENT SOURCES OF FUGITIVE DUST (FINDING #11)

Uncertainties in fugitive dust emission estimates for crustal materials can be estimated and reduced to acceptable ranges by reconciliation with ambient measurements. Variations in particle size and chemical composition of the fugitive dust at a receptor site as well as the temporal and spatial variations for multiple receptor sites can be used to indicate the spatial scale of the sources, the portion of the day when fugitive dust contributions are large, and whether the fugitive dust is wind generated. Additional chemical and physical measurements of the ambient aerosol at receptor sites may shed some light on specific dust sources. Receptor measurements should be used with model estimates to evaluate modeled source contributions and to focus inventory improvement efforts.

12.1 Technical Support of Finding #11

Comparisons between source contributions estimated by receptor models and emissions inventories have called attention to and have assisted in improving fugitive dust inventories. Figure 12-1 illustrates the current discrepancy between emissions models and receptor-measured source contributions to $PM_{2.5}$ in ambient air determined by receptor models. It compares fractional contributions from different particle sources derived from a state-of-the-art emissions inventory and average source contributions calculated from the Chemical Mass Balance (CMB) receptor model (Watson et. al., 1998). CMB fractions are calculated with respect to the total $PM_{2.5}$ mass and with respect to that from primary source types. Secondary ammonium nitrate and ammonium sulfate form from gaseous oxides of nitrogen, sulfur dioxide, and ammonia emissions that are separate from the $PM_{2.5}$ inventory. Even when these species are removed from the total $PM_{2.5}$, there are substantial discrepancies. The most notable discrepancy is that of fugitive dust, with the inventory showing nearly twice the fraction accounted for by the receptor model apportionment.

Figure 12-1. Comparison between fractional source $PM_{2.5}$ contributions estimated by CMB receptor model and emissions inventory for Denver during 1996 (Watson et. al., 1998).



Receptor models use the variability of chemical composition, particle size, and concentration variations in space and time to identify source types and to quantify source contributions. Commonly-applied and emerging receptor models include:

Chemical Mass Balance (CMB). Ambient chemical concentrations are expressed as the sum of products of species abundances and source contributions. These equations are solved for the source contributions when ambient concentrations and source profiles are supplied as inputs. Several different solution methods have been applied, but the effective variance least squares estimation method is most commonly used because it incorporates precision estimates for all of the input data into the solution and propagates these errors to the model outputs. The CMB model provides the basic structure for other pattern based receptor models, and they can all be derived from the CMB equations.

Enrichment Factors (EF). Ratios of atmospheric concentrations of chemical components to a reference component are compared to the same ratios in geological material that represents an average crustal composition for a region or sampling site. Differences are explained in terms of other sources. Heavy metal enrichments are usually attributed to industrial emitters. Sulfur enrichment is attributed to secondary sulfate. Potassium enrichment is attributed to burning and cooking. Soil and road dust compositions may differ from regional crustal compositions, and enrichments or depletions for certain species may indicate the presence or absence of more or less of these non-continental dust contributions.

Multiple Linear Regression on Marker Species (MLR). Mass or crustal concentrations are expressed as a linear sum of unknown regression coefficients times source marker concentrations measured at a receptor. The markers must originate only in the source type being apportioned. This is a stringent assumption that is rarely met in practice, especially for suspended dust. The regression coefficients represent the inverse of the chemical abundance of the marker species in the source emissions. The product of the regression coefficient and the marker concentration for a specific sample is the tracer solution to the CMB equations that yields the source contribution.

Temporal and Spatial Correlation Eigenvectors. Principal Component Analysis (PCA) and Factor Analysis (FA) are in this category. For a long time series (several hundred samples), temporal correlations are calculated from a time series of chemical concentrations at one or more locations. Eigenvectors of this correlation matrix are determined and a subset is rotated to maximize and minimize correlations of each factor with each measured species. The factors are interpreted as source profiles by comparison of factor loadings with source measurements. Several different normalization and rotation schemes have been used, but their physical significance has not been established. For a large spatial network (25 to 50 locations located along known or anticipated concentration gradients), spatial correlations are calculated from chemical measurements taken on simultaneous samples at a large number of locations. Eigenvectors of this correlation matrix represent a spatial distribution of source influence over the area, providing that the samplers have been located to represent the gradients in source influence. As with temporal correlation models, several normalization and rotation schemes have been applied.

Neural Networks. Known inputs and outputs are presented to a neural network that simulates the human thought process. The network assigns weights to the inputs that reproduce the outputs. Once these patterns have been established for cases where outputs are known, the weights can be applied to input data to estimate outputs. Neural networks can also be used to provide function relationships and represent a solution to the CMB equations.

12.1.1 Chemical Composition

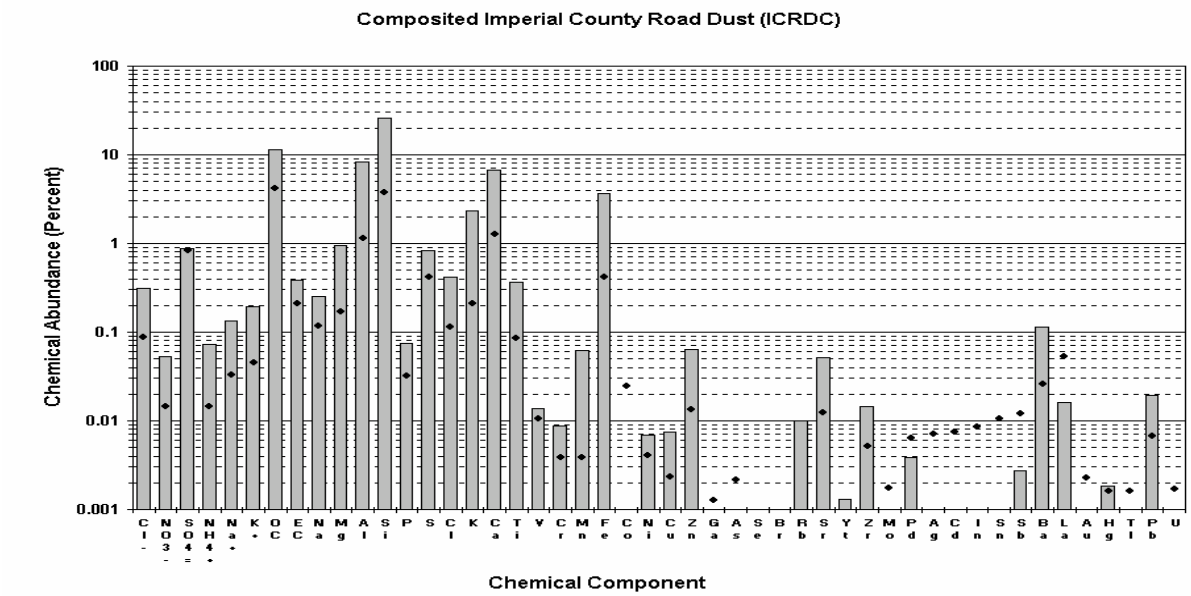
These methods typically require some knowledge of the fractional mass abundances of measured properties and the uncertainties of those abundances for specified particle size fractions, commonly termed the source profile. These profiles can be determined by sampling and analysis of dust that is believed to be a contributor or, under some circumstances, they can be derived from the ambient data themselves. Several of the multivariate methods described above intend to draw these profiles from the ambient measurements. For quantitative source apportionment, source profile properties must meet the following criteria:

- their abundances (mass fraction) are different in different source types;
- their abundances do not change appreciably during transport between source and receptor (or such changes can be simulated by measurement or modeling); and
- their abundances are reasonably constant among different emitters and operating conditions for a selected source type.

Hundreds of geological samples from paved and unpaved roads, construction sites, industrial operations (e.g., cement, minerals handling, mining), and native surfaces have been taken, suspended, sampled through TSP, PM_{10}, or $PM_{2.5}$ inlets, and analyzed for elements, ions, and carbon fractions (Chow et. al., 1994, Chow and Watson, 1994). Most of these profiles represent urban sources, and few are available for areas surrounding IMPROVE sampling sites or from the large land masses between urban emitters and Class I areas.

Figure 12-2 shows a typical geological profile for a paved road dust sample. Silicon is nearly always the most abundant component, typically constituting 9% to 25% of the mass. Aluminum is usually 5% to 15% and is often enriched in the $PM_{2.5}$ fraction compared to the coarse fraction owing to a greater quantity of aluminum oxides in clay. Iron typically constitutes 3% to 10% and calcium constitutes 5% to 20% of most crustal material. Carbon, measured as organic (OC) or elemental (EC) constitutes 5% to 20% of paved road dust and agricultural soils, but is typically less than 5% in desert soils. Potassium, manganese, and magnesium are usually found in oxide forms to the extent of a few percent in most soils. Except where amendments are applied or in dried salt pans, most of the crustal potassium is insoluble, as illustrated in Figure 12-2 where only 10% is measured in its ionic form (K^+). Other elements may be present depending on the nature of the underlying geology or due to contamination from other sources. The road dust sample depicted in Figure 12-2 shows evidence of contamination from nickel, copper, zinc, barium and lead, all typical of residue from vehicle abrasion, exhaust system deterioration, and use of fuel additives such as barium and lead (Note: leaded fuel was used in vehicles coming from Mexico when the road dust sample was characterized in 1993).

Figure 12-2. Example of a PM_{10} paved road dust source profile from Imperial County, CA (Watson and Chow, 2000). The height of each bar represents the average abundance for samples from several roads and the diamond represents the standard deviation of the average abundance.



Differences among the major soil elements are usually too small to allow different sources to be distinguished by receptor models. The variability of the non-recurring elements is usually too high for them to separate one source from another. These observations typically apply to urban situations where the underlying geological strata is often similar throughout the study area. The desert southwest contains many exposed strata (evident from their different colors) that may have a large enough variation in composition between strata, and a small enough variability between strata, to develop profiles that could be separated by a receptor model.

The National Soil Characterization database is being compiled to document the information listed in Table 12-1, where and when it is available (http://www.statlab.iastate.edu/ soils/ssl/natch_data.html). This information is for bulk soil samples that do not take into account the fact that compositions are different for different particle sizes. There is also little overlap between the quantities measured in Table 12-1 and the profile species illustrated in Figure 12-1. This data base might be used, however, to design a systemic source testing program that would acquire samples from different geological formations near and surrounding Class I areas. A set of profiles could be developed for PM_{10} and $PM_{2.5}$ that would determine the extent to which contributions from different geological strata might be characterized.

Although elements, ions, and carbon are most commonly measured, Watson and Chow (2000) identify several other characteristics of geological material that might distinguish among different sources. Additional chemical and physical properties beyond the commonly measured elements, ions, and carbon need to be examined to determine which ones can meet the criteria required for practical source apportionment applications. Several types of analyses that might meet these needs are:

Appearance: Optical microscopy provides a semi-quantitative overview of the shape, size, mineralogy, and type of different particles with geometric diameters larger than ~2 μm (Casuccio et. al., 1983, 1989; Crutcher, 1982; Daves et. al., 1993; Draftz, 1982; Eswaran, 1972; Grabowska-Olszewska, 1975; Grasserbauer, 1978; van Malderen et. al., 1996; Zou and Hooper, 1997).

Table 12-1. Types of data being acquired in the National Soil Characterization database.

KIND ANAL	AMT SUB	KIND ANAL	AMT
39= PS	5 PS, FRAGMENTS UNSPECIFIED	48=	-ELECTRICAL CONDUCTIVITY, RESISTIVITY, SALT
	6 PS, FRAGMENTS MEASURED	49=	-SATURATION EXTRACT SOLUBLE, CATIONS & ANIONS
	7 PS, FRAGMENTS ESTIMATED	50=	-EXTRACTABLE IRON (PYROHOS, DITH-CIT, OXALATE)
	8 PS, FRAGMENTS MEAS. & EST.	51=	-CATION EXCHANGE CAPACITY
40= BD	5 BULK DENSITY WITHOUT COLE	52=	-ALUMINUM (PYROPHOS, KOH, DITH-CIT, NAOH, OXALATE)
	6 BULK DENSITY WITH COLE	53=	-ATTERBERG LIMITS
41= MIN	5 CLAY FRACTION	54=	-CLAY: CARBONATE OR FINE
	6 SAND A/O SILT FRAC(S)	55=	-FIBER
	7 CLAY,SAND A/O SILT FRAC(S)	56=	-GYPSUM (TOTAL)
	8 FAMILY MINERALOGY	57=	-MINERAL CONTENT ORGANIC SOIL
42= HL	5 STANDARD ITEMS (MECHANICAL ANALYSIS & ATTERBERGS)	58=	-MANGANESE (ANY METHOD)
	6 STANDARD ITEMS & MAXIMUM DENSITY	59=	-NITROGEN
	7 STANDARD ITEMS, MAXIMUM DENSITY, & SHRINK-SWELL	60=	-PHOSPHOROUS (TOTAL)
		61=	-SULFUR (TOTAL)
		62=	-FIELD MEASURED WATER CONTENT OR TABLE, PERK RATE, ETC.
43=	- WATER CONTENT OR TENSIONS	63=	-THIN SECTIONS
44=	- ORGANIC CARBON OR MATTER	64=	-TOTAL ELEMENTAL ANALYSES
45=	- PH	65=	-HYDRAULIC CONDUCTIVITY
46=	- EXTRACTABLE CATIONS/BASES	66=	-AVAILABLE PHOSPHOROUS
47=	- CARBONATE	67=	-AVAILABLE POTASSIUM1

Soluble and insoluble inorganic elements and compounds: X-ray fluorescence (Watson et. al., 1999), proton-induced x-ray emission (Cahill et. al., 1981), instrumental neutron activation analyses (Ambulkar et. al., 1993, 1994; Weginwar and Garg, 1992), atomic absorption spectrophotometry (Chow et. al., 2000), inductively coupled plasma-mass spectrometry (Adetungi and Ong, 1980; Davis et. al., 1984; Davis and Chen, 1993; Esteve et. al., 1997), computer automated scanning electron microscopy (Cassucio et. al., 1983, 1989; Grabowska-Olszewska, 1975), ion chromatography (Chow and Watson, 1999; Tabatabai and Dick, 1979), and automated colorimetry have been applied, sometimes in conjunction with different extraction and selective sample preparation techniques, to quantify a variety of elements, water soluble components, 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 (Higashi and Flocchini, 2000; Meuzelaar, 1992).

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 (Bomboi et. al., 1990; Chuang et. al., 1995; Rogge et. al., 1993; Ruiz et. al., 1998; Simoneit, 1977, 1979; Simoneit et. al., 1988; Weir and Ireson, 1989; Wild et. al., 1990).

DNA, toxins, microbes, and bacteria: Biologically specific tests can be applied to characterize these substances (Bolton et. al., 1985; Bowman et. al., 1991; Bruns et. al., 1998; Fraser et. al., 1988; Krahmer et. al., 1998; Lechevalier, 1977; Simoneit, 1979; Song et. al., 1999; Vestal and White, 1989; Zelles et. al., 1992).

Isotopes: Low-level radioactive counting, dilution mass spectrometry, and accelerator mass spectrometry are potential methods to quantify unique isotopic abundances in geological materials (Biscaye et. al., 1974; Bunzl et. al., 1994; Clayton et. al., 1972; Douglas and Savin, 1975; Goh et. al., 1977; Grousset et. al., 1988; Harrison et. al., 1993; Hirose and Sugimura, 1984; Jackson, 1981; Jackson et. al., 1982; Kerley and Jarvis, 1997; Kiefert and McTainsh, 1996; Kusumgar et. al., 1980; Le Roux et. al., 1980; Nho et. al., 1996; Novák et. al., 1996; Patterson et. al., 1999; Sridhar et. al., 1978; Turekian and Cochran, 1981).

Although all of these characteristics have been applied to different crustal samples at different times, as indicated by the cited references, they have not been applied in conjunction with the more commonly available elemental, ion, and carbon measurements made on previous crustal and ambient samples. The degree to which they can distinguish between different fugitive dust source types, and the relevance of those source types to dust contributions in Class I areas, needs to be evaluated by a critical review of previous studies. Where appropriate, these should be incorporated into a comprehensive source test program.

12.1.2 Particle Size

As noted earlier, the majority of suspended geological dust is in the coarse particle size fraction, but even in the $PM_{2.5}$ fraction its particle sizes are weighted toward larger particles (i.e. between 1 and 2.5 µm) as opposed to other components such as sulfate and nitrate that have a peak distribution in the 0.2 to 0.7 µm size range. Cahill et. al. (1987) have taken measurements with the DRUM impactor for many years at IMPROVE sites and have reported substantial differences in crustal indicator (e.g. aluminum, silicon) ratios and absolute amounts between different impactor stages. These data could be further analyzed to identify differences in composition and particle size that might indicate source origins. The coarse mass to $PM_{2.5}$ soil mass ratio in the IMPROVE data base might provide some indication of the zone of influence for different sources. Low ratios are consistent with distant sources while high ratios would indicate a higher local contribution.

12.1.3 Temporal and Spatial Variability

The value of temporal variability in identifying the quantity of dust contributed at a certain time, the contribution of this short-duration contribution to the 24-hour average, and possibly even the direction of the source when coupled with corresponding wind direction measurements, is

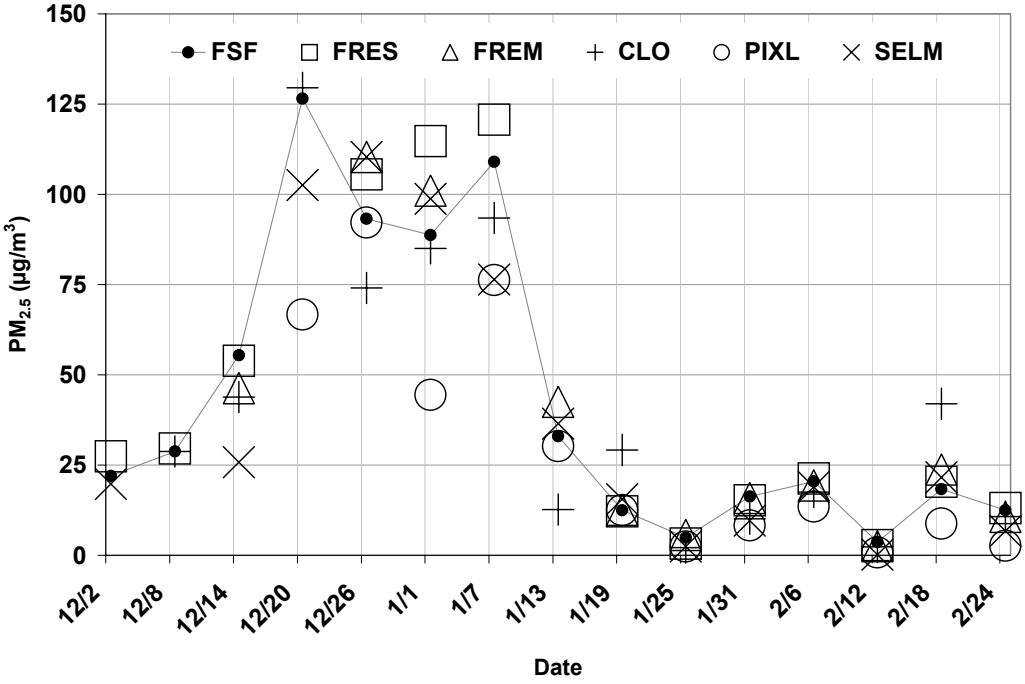
demonstrated in the coarse particle time series depicted in Figure 7-3. Unfortunately, collocated BAM or TEOM monitors may not be possible at many sites. Several studies have been conducted that locate portable, battery-powered samplers around a long-term measurement site, or close to sources that might affect a long-term measurement site, to better understand contributions from nearby sources in relation to contributions of sources that affect neighborhood and urban areas. The configuration of a typical satellite site, consisting of two Minivol samplers and a Radiance nephelometer equipped with a smart heater (discussed below), is illustrated in Figure 12-3.

Figure 12-3. Satellite site configuration at the Fresno residential site (Watson et. al., 1998, 2000. The two Minivols are on the left and the Radiance nephelometer with smart heater is in the box on the right. These are supported by a power drop from the power pole, but they can also operate from batteries and a small solar panel in remote locations.

Figure 12-4 compares 24-hour $PM_{2.5}$ measurements from five satellite sites using Minivols with measurements made at the Fresno, CA "Supersite." FSF, the "Supersite" location, appears to represent $PM_{2.5}$ over the urban area, although there are spatial deviations at nearby sites during high $PM_{2.5}$ episodes. The FRES residential site shows higher $PM_{2.5}$ than FSF on December 26, January 1, and January 7. This is believed to be caused by residential wood burning at homes near FRES. $PM_{2.5}$ at FREM, which is next to a freeway on-ramp and is affected by vehicle exhaust and road dust, deviates by no more than ±15% from $PM_{2.5}$ at FSF on a few days, and is nearly identical on most days. Increments in light scattering over the non-urban PIXL site and the downwind SELM site are clear during high $PM_{2.5}$ episodes. Similar $PM_{2.5}$ concentrations are found at all sites for the lower values recorded from January 19 through February 24, indicating a dominance of regional-scale as opposed to local-scale contributions. $PM_{2.5}$ concentrations at CLO are least related to values at FSF or at any of the other sites. Chemical characterization of

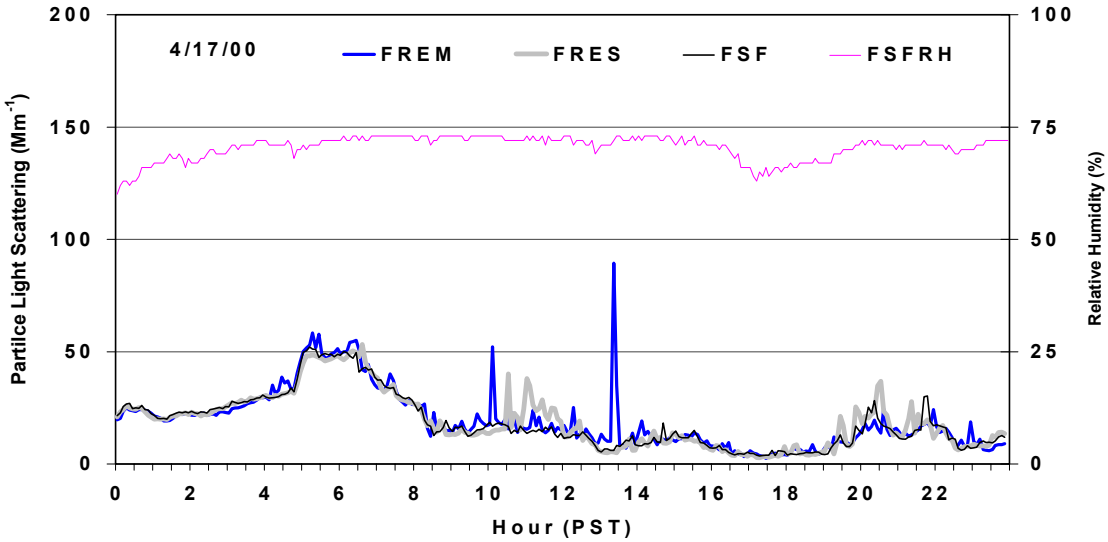
these samples will indicate the extent to which fugitive dust contributes relative to other chemical components such as carbon and secondary sulfates and nitrates.

Figure 12-4. Spatial distribution of 24-hour average $PM_{2.5}$ around the Fresno "Supersite" (FSF) during winter 1999/2000. $PM_{2.5}$ at the FSF site is measured with a sequential filter sampler and $PM_{2.5}$ at the other sites are measured with Minivols. Other sites include residential sites 0.5 km E (FRES) and 5 km NE (CLO), a site near a freeway onramp 1 km W (FREM), a boundary site at the Selma Airport ~25 km S of Fresno (SELM), and a rural site in the Pixley Wildlife Refuge (PIXL)100 km S of FSF.



Radiance® nephelometers measure light scattering that is an indicator of particle concentration. Dust has a lower light scattering efficiency than smaller particles, but known dust events do register on this nephelometer. The nephelometer is equipped with a "smart heater" that raises the temperature of the air stream when relative humidities rise above 70% to maintain it at less than 70% RH. This heating minimizes interferences from liquid water absorbed by soluble particles while retaining volatile particles such as ammonium nitrate that evaporate at temperatures exceeding ~15 °C. Figure 12-5 compares nephelometer responses from three of the nearby sites shown in Figure 12-4 (i.e., sites FSF, FRES, and FREM). All of these traces track the urban-scale changes throughout the 24-hour period. Even though the FRES and FREM monitors are separated by more than 1 mile, the general diurnal pattern appears as if they were right next to each other. Short-duration spikes show the effects of local emissions that affect one monitor but not the others. At the FREM motor vehicle site these represent contributions from nearby road dust or vehicle exhaust. Over time, these spikes can be plotted with wind direction to determine the direction of their origin.

Figure 12-5. Radiance® nephelometer five-minute average readings of particle light scattering at the Fresno "Supersite" (FSF), the motor vehicle site (FREM) and the residential site (FRES). Relative humidity is also plotted to show how the smart heater maintains RH at ≤70% to minimize scattering from liquid water and volatile particle losses.



12.2 Limitations

The application of receptor models to estimating fugitive dust emissions is limited due to lack of measurements. The most important deficiency is the lack of source profiles for the land forms that are probable contributors in the ambient PM_{10} and $PM_{2.5}$ fractions for Class I areas in the western US..

12.3 Information Gaps

Additional chemical and physical properties need to be examined to determine which ones can meet the criteria needed for practical source apportionment applications. The detailed source profiles obtained will be used to identify those components in source emissions that are likely to be detectable in ambient air and have the potential to distinguish the contributions of different types of dust emitters to ambient PM_{10} and $PM_{2.5}$ concentrations.

12.4 Recommendations

Examine the IMPROVE data base for enrichments of geological elements relative to median ratios or to soil compositions typical of the areas in which samplers are located. Determine the extent to which different elemental ratios are observed, outside of natural variability, and the extent to which these are correlated with elevated $PM_{2.5}$ soil or coarse particle mass concentrations.

Examine DRUM impactor measurements for soil related species from IMPROVE special studies. Identify typical size distributions and deviations from those distributions. Relate outliers to $PM_{2.5}$ and coarse mass concentrations.

Examine nephelometer light scattering data from the IMPROVE network. Determine the extent to which spikes are observed and whether these can be related to fugitive dust or other sources.

Critically review previous studies (cited above) of microscopic, isotopic, organic, and other measurement methods with respect to their ability to distinguish among different land forms and fugitive dust emitters. Determine which methods can be practically applied to source characterization and ambient sampling.

Design and implement a systematic source profile measurement program. Examine existing soil and geological maps to identify areas that have suspension potential. Obtain sufficient dust quantities from these areas for a variety of analysis methods; suspend and sample these quantities through $PM_{2.5}$ and PM_{10} size-selective inlets; analyze by the selected methods; and apply appropriate mathematical tests to determine the degree to which profiles are collinear or are dissimilar enough to distinguish between fugitive dust source types and areas.

Implement high time resolution methods, preferably ones that are specific to coarse particles, at several IMPROVE sites and between these sites and suspected source areas. Examine these to determine when dust events occur, their durations, and their contributions to 24-hour averages. Use this information to focus emissions inventory efforts.

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