A Review of Methods for Measuring Fugitive PM-10 Emission Rates From Stationary Sources

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Abstract

The purpose of this report is to serve as a guide for the measurement of fugitive dust from stationary sources. To that end, the methods of measuring fugitive particulate emissions are reviewed. The methods included are the quasi-stack method, the roof monitor method, the upwind-downwind method, the exposure profiling method, the portable wind tunnel method, the scale model wind tunnel method, the tracer method and the balloon method. Each measurement method is explained, along with its advantages and disadvantages. Sources of error are discussed, as are sampling protocols. The literature on each method is reviewed. A section of the report is devoted to the issues of error, accuracy and precision of the methods.

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Introduction

Fugitive dust is dust emitted from sources other than stacks. EPA now regulates emissions of dust particles which have an aerodynamic diameter of ten microns or less, because this dust causes respiratory health effects. Such dust is referred to as PM-10.

Emission factors published in EPA document AP-42 describe fugitive dust emission rates for a variety of sources. Most of the time these emission factors suffice for calculation of industrial or other fugitive emissions. But sometimes people in the private sector or state or local government disagree with the published emission factors for a given process or situation, or they think that the published emission factors do not apply. They wish to calculate specific emission factors themselves. In that event, the rate of fugitive dust emission must be measured. The purpose of this report is to provide information and guidance about the measurement of PM-10 from fugitive sources. To that end, a review of the literature concerning methods for measuring fugitive PM-10 emissions has been performed.

Several such methods exist. The quasi-stack method, the roof monitor method and the upwind-downwind method have relatively long histories, and have been used to measure various kinds of fugitive emissions including dust. The exposure profiling method was developed specifically for measuring fugitive particulate emissions. The portable wind tunnel method was first used by soil scientists before being used in an air pollution context. The balloon method is a little-used offshoot of the exposure profiling method. The scale model wind tunnel method and tracer method have also been comparatively little-used.

The selection of a measurement method depends upon such factors as source geometry, presence or absence of an enclosing structure, feasibility of hooding or enclosing the source, size of the dust plume, distance between plume generation and feasible sampling sites, and type of process causing the plume. For example, the quasi-stack method requires the (usually temporary) enclosure or hooding of a source. The roof monitor method involves monitoring of air flow and particle concentration leaving all major exit points in a building. The portable wind tunnel is used only to study emissions from wind erosion. Exposure profiling is an excellent method for studying "point" sources such as loading or unloading operations, or "line" sources such as traffic on a road, but the sampling equipment must be placed within a few meters of the emission source. The upwind-downwind method is nearly universally applicable, but may be the least accurate of the methods. Appendix K (TRC, 1980) contains excellent information on the selection of a measurement method.

Quasi-stack method

Richards and Brozell (1992), Richards and Kirk (1992), and Brozell and Richards (1993) describe recent applications of the quasi-stack method at stone crushing plants. The quasi-stack method is especially well suited to small materials-handling operations and small components of industrial processes. Essentially, this method consists of enclosing or hooding (often temporarily) the fugitive dust source to be measured. The dust plume is ducted away from the source at a known air velocity, by using a fan, and the exhaust is sampled isokinetically in the duct.

The intake velocity must be lower than the velocity in the sampling duct. For typical ducts with smooth walls the Reynolds number should be in the neighborhood of 200,000 (turbulent region). There should be a minimum straight duct run of three duct diameters upstream and downstream of the sampling port (Kolnsberg et al, 1976).

Standard stack sampling trains (EPA Methods 201 or 201A) may be used to measure concentrations of PM-10, using standard sampling protocols (EPA Method 1, where applicable). The product of the concentration, the mean velocity of the exhaust and the cross-sectional area of the duct gives the emission rate.

The quasi-stack method is potentially the most accurate means of measuring a fugitive dust plume because the entire plume is captured and measured close to the source, and because it uses well established and well validated sampling protocols. However, the air velocity in the vicinity of the hood or enclosure must be sufficient to entrain the entire PM-10 plume without being fast enough to cause excess emissions.

For example, excess emissions might emit from a stone crusher if the air speed inside the temporary enclosure is higher than the normal ambient air speed. In that case, the higher air speed in the enclosure might cause more dust to enter the air from stone crushing, thus causing an overestimation of the emission rate.

Also, there must not be significant deposition of PM-10 within the duct-work or enclosure. Furthermore, if the space enclosed is normally subjected to turbulence from ambient winds, the emission rate calculated after enclosure may underpredict the true emissions. Finally, the sampling protocol must represent the average dust levels encountered in cyclic or uneven dust-producing processes (Cowherd and Kinsey, 1986).

Appendix A is an excerpt from 40 CFR 51 containing descriptions of Methods 201 and 201A. Appendix A also contains excerpts from 40 CFR 60, with descriptions of Methods 1 and 5D. Appendix B is an excerpt from Richards and Brozell (1992) describing recent applications of the quasi-stack method.

EPA published a series of technical manuals on measuring fugitive emissions in 1976. One manual was on the quasi-stack method (Kolnsberg et al), one was on the roof monitor method (Kenson and Bartlett) and one was on the upwind-downwind method (Kolnsberg). From the point of view of measuring PM-10, these manuals have several problems: they are old, the equipment in them has largely been superseded, the manuals were written from the perspective of measuring all fugitive emissions, not just dust, and at that time EPA was concerned with measuring total suspended particulate, not PM-10. Nevertheless, they provide significant useful information and are being included in this report as appendices. However, it must be reiterated that much of the equipment in these manuals has been superseded. Appendix C contains the text of Kolnsberg et al, (1976), the manual on the quasi-stack method. Appendix K (TRC, 1980) also contains very detailed information on this method, although the equipment described is out of date.

Some specific work has been done on hood capture of process fugitive particulate by PEDCo Environmental, Inc. (1984) and by Kashdan et al (1986). The former study describes the capture of fugitive particulate from a primary copper convertor by use of an air curtain, and the use of quasi-stack measurements to quantify emission rates. There is very good documentation of adequate capture efficiency of this arrangement, but no documentation that the fugitive emissions are unaffected by the air curtain. Nevertheless, the air curtain is quite far from the process, and it seems likely that the very small negative pressure involved would be too small to cause increased emissions. The air curtain seems useful only for heated, buoyant plumes.

Kashdan et al comprehensively describe a series of hood designs for capture of process fugitive particulate emissions. Capture efficiencies are included. Again, however, there is no information available on the extent of influence of these hood systems on the processes themselves. To what extent do they induce increased emissions? Could they reduce emissions by decreasing turbulence around the source? Obtaining answers to these questions is not necessarily a trivial problem.

Richards and Brozell (personal communication, 1993) have used a smoke tracer method to visually determine the minimum air velocity required for PM-10 plume capture. This issue is further complicated if ambient winds or drafts must be dealt with, because the hood air velocity needs to be higher in draftier environments (Kolnsberg et al, 1976). Also, it must be ascertained that the behavior of the visible smoke plume resembles that of the actual PM-10 plume. Furthermore, it would be preferable to have mass measurements of emitted and captured tracer as well as the visual evidence that the hood is effective at capturing emissions without inducing or decreasing them.

In any case, several hood designs may be appropriate for use with quasi-stack measurements. The user must demonstrate, however, that the hood does not cause underestimation or overestimation of source emissions.

Roof monitor method

When processes are located within a building, the roof monitor method may be the best means of measuring fugitive particulate emissions. In this method, measurements of particulate concentration and air velocity must be made at each opening from which dust may issue from the building. The cross-sectional area of each opening is also required. The product of the cross-sectional area of the opening, the exit velocity, and the concentration of PM-10 gives the fugitive PM-10 emission rate from an opening. The sum of the emission rates from all openings gives the emission rate for the building as a whole.

In most cases, the building as a whole is considered to be the "source." When considering the ambient impact of processes within a building, we are only interested in dust which escapes from the building, rather than in the "true" emissions from each process inside.

Air velocity in openings to buildings may be quite variable. Even flow direction may shift. Consequently, isokinetic sampling may be difficult, and it may not be feasible to use stack testing methods. In that event, ambient PM-10 sampling devices may be used. These devices may pump a measured flow of air past a filter. The weight of particulate deposited divided by the total air flow during the time the device was in operation gives the average concentration of dust in the sampled air. The product of the average concentration, the cross-sectional area of an opening, and the average exit velocity will give an average emission rate for a given opening over the period of time sampled. Appendix D contains a list of ambient samplers which have met EPA criteria published in 40 CFR 50, as of July, 1993. Table I (from Muleski et al, 1991) provides a list of advantages and disadvantages of various types of PM-10 ambient samplers.

Another issue when using the roof monitor method is that concentrations of dust may vary in unknown ways across various openings. Consequently, it is important to sample, as in stack testing, at a number of sites along the cross section of each opening.

In cases where ducts lead to openings, it is important to ascertain that there is not significant PM-10 deposition in the duct-work downstream from the sampling site before the exit from the building is reached. Otherwise one will make significant overestimations of PM-10 emissions.

On the other hand, it is critical to sample during times which are representative of normal and peak dust emissions. Otherwise, the calculated emission rates will have little meaning.

Without the use of additional testing, it will not be possible to separate and quantify the individual sources within a building; the different plumes will be measured as one intermingled plume leaving the various openings of the building. To discriminate between sources under one roof, tracer tests are required (see Appendix E, and also see Vanderborght et al 1982), or else one process at a time may be operated to obtain an emission rate for each process.

The roof monitor method should have the potential to give accurate emission rates. It has been thought to be somewhat less accurate than the quasi-stack method, however (Kolnsberg, 1982). Another issue that may arise in sampling via the roof monitor method is that the building openings may be difficult to access, difficult or hazardous to lead electrical lines to, and precarious to work around. Trozzo and Turnage (1981) developed a protocol for using battery powered personal samplers as surrogates for the large hi-vol ambient samplers which were then the EPA reference method for measuring ambient dust concentrations. No subsequent studies using this technique were found in the literature. Newer battery powered devices called saturation monitors could be adequate under some conditions for the roof monitor method, but this has not been studied. Generally, if stack sampling methods cannot be used, it is recommended that EPA approved ambient sampling devices be used whenever possible (See Appendix D).

However, it is EPA's recommendation that whenever feasible, stack sampling trains be used, specifically Method 201 or 201A. It may be desirable to build temporary duct-work around openings in order to use these methods, provided that the duct-work does not alter the dust outflow.

In the case where emissions are sampled in ducts, EPA Method 1 should be used when the ducts are of the appropriate type. In cases where sampling is attempted in an actual roof monitor, the sampling should be done according to EPA Method 5D. (See Appendix A.)

Appendix E contains the 1976 technical manual on the roof monitor method by Kenson and Bartlett. As noted above, there is substantial obsolete material in this manual; we include it nevertheless because there is also substantial valuable information. Appendix K (TRC,1980) also has detailed information on the roof monitor method (but dated information on equipment).

Upwind-downwind method

In the upwind-downwind method, at least one ambient PM-10 concentration is obtained upwind of a dust source and several PM-10 concentrations are obtained downwind as well. Wind speed and direction and other meteorological variables are monitored during the sampling procedure. The downwind concentration minus the upwind concentration is considered to be the concentration due to the PM-10 source (or net concentration). Using a dispersion model and the meteorological information, the net concentration is used to solve for the emission rate in the dispersion model. Each downwind sampler will yield an emission rate estimate; these may be averaged to obtain the best estimate of the emission rate.

The upwind-downwind method may be applied to many different situations. It cannot, however, distinguish between plumes which mix, unless one of the plumes is distinctly upwind of the other. While the upwind-downwind method is the most versatile of the generally applied methods, it is also been considered the least accurate. This is partly because only a tiny fraction of the greatly diluted plume is sampled, and this sampling is usually done many meters from the source. While plumes are thought to behave in a Gaussian fashion, that behavior occurs only on average over a period of time. A great many samples over a long time would have to be obtained for the actual plume distribution to approach that of a Gaussian curve. Consequently, random plume irregularities may give rise to inaccurate emission estimates.

Even if sampling is done at many sites (an expensive proposition), inaccuracies still result from using average meteorological values to represent the instantaneous vagaries of real weather. For example, the dispersion models are particularly unable to cope with a situation in which the wind direction at the source is different from the wind direction at the receptor.

Despite these problems, it seems possible to obtain reasonable accuracy with this method. Hu Gengxin et al (1992) found that their results were within a factor of two, 80 percent of the time, apparently using the quasi-stack method as a reference.

In any case, there is an important reason for using the upwind-downwind method: there are times when this is the only method which suits the situation. Obtaining an emission rate from an area source such as a large parking lot is an example.

Regarding basic sampling protocol, the arrangement of sampling devices will vary depending upon the geometry of the source. The number of upwind samplers will depend upon the proximity of interfering upwind plumes--a more heterogeneous upwind dust profile will require more upwind samplers. Downwind of the source to be measured, for "point" or area sources, at least five ambient particulate samplers are required, at two different downwind distances and three different crosswind distances (Cowherd and Kinsey, 1986). The greater the number of downwind samplers, the better the characterization of the plume. Refer to Appendix D for a list of acceptable ambient sampling equipment, and for an excerpt from the statute which defines the reference method for measuring PM-10 in ambient air.

Kinsey and Englehart (1984), Russell and Caruso (1983), Maxwell et al (1982), and Larson et al (1981) have done upwind-downwind studies on "line" sources (roads). However the exposure profiling technique is well suited to roads, and is thought to be more accurate than the upwind-downwind method (Kolnsberg,1982; Fitzpatrick, 1987).

Looking at sampling arrangements in more detail, a study by

Carnes et al (1982) suggested that 10 or 11 downwind samplers was the optimum number for measuring emissions from a coal storage pile, based upon a cost-benefit analysis. They claimed that using ten downwind samplers will provide estimates of emission strength within 25 percent of estimates obtained using 30 or more samplers. Hesketh and Cross (1983) make no specific recommendations on total number of samplers, but do suggest two sampling heights for each sampling site, one at ground level and one at three meters. Axetell and Cowherd (1984) did an exhaustive study on surface coal mines; they wrote in detail on most of the measurement methods described in this report, including the upwind-downwind method. Excerpts of their report are included as Appendix F. The reader should keep in mind, however, that the equipment in that study was used primarily to measure total suspended particulate, not PM-10. Appendix K (TRC, 1980) also contains a good deal of information on the upwind-downwind method. Kolnsberg (1976) wrote a technical manual on the method. That report is included as Appendix G because of its valuable detail, despite the obsolescence of much of the equipment described.

Regarding equipment, some studies (Kinsey and Englehart, 1984; Russell and Caruso, 1983; and Larson et al, 1981) have used devices which turn off the ambient samplers automatically if the wind direction deviates more than a certain number of degrees from the source. This is done because the sampler may be essentially out of the plume if the wind deviates enough. Shut-off angles for these devices have typically been in the range of 22.5 - 65 degrees to either side of the original plume centerline. The desirable shut off angle will vary with the distance the samplers are from the source. Other studies (Maxwell et al, 1982; Carnes et al, 1982; Larson, 1982; and Wells et al, 1980) have not used such a device. Current thought is that using an automatic shut-off is a good idea (Cowherd, C., 1993, personal communication). Hesketh and Cross (1983) suggest using two ambient samplers at each sampling position, one operating continuously and the other operating only when the wind is within 22.5 degrees from the source. Any sampler with a directional shut-off should have a timer to count the elapsed time the sampler is in operation.

Factors other than wind direction changes may make the data from a particular test run unusable. For example, if the wind is very slight, a recognizable plume might not form. A typical response has been to initiate testing only if wind speeds exceed 1 meter per second (2.2 m.p.h.).

Another important issue relevant to the upwind-downwind method is the choice of a dispersion model. Which model should one use?

EPA uses the Industrial Source Complex (ISC) model, particularly for gaseous emissions. This is a Gaussian plume model for flat terrain. It has no deposition term for particles under 30 microns in aerodynamic diameter, meaning that it does not account for deposition of these particles downwind of the source. PM-10 will have some degree of downwind deposition. So the ISC model will tend to overestimate PM-10 concentrations downwind for a given emission rate, or if using the upwind-downwind method, the model will underestimate the emission rate for a measured downwind concentration.

The rate of downwind deposition will depend upon air convection and turbulence which bring particles into contact with the ground, and upon the gravitational settling velocity of the particles. The gravitational settling flux and ground deposition flux are both thought to be proportional to the local air concentration of particles (Ermak, 1977). EPA is nearing completion on work to add a deposition term to the ISC model, which should make it more accurate for use with dust.

There are other dispersion models available which have deposition terms. Ermak (1977) developed a model based upon the solution of an atmospheric diffusion equation. Several later models are based upon his work. These include models developed by Winges (1990 and 1982), and by Becker and Takle (1979). Winges's Fugitive Dust Model (1990) has computer software which allows non-scientists to perform the data entry.

Hu Gengxin and Yang Xu (1992) reported on the development of a model by Hu Gengxin and Xia Liguo. Hu Gengxin et al (1992) briefly reviewed the applicability of various dispersion models to fugitive dust problems, and compared a model developed by them to two previously developed by Hu Gengxin. They used known emission rates to evaluate the models, and found that their new model performed somewhat better overall than Hu Gengxin's older ones. They also found that each model had optimal distances and angles from the plume centerline where it performed better than the other models .

Generally, when using dispersion models, at a minimum the following information will be required: Distance from each ambient sampler to dust source, wind speed, wind direction, and Pasquill-Gifford or Pasquill-Turner stability class. Other parameters, such as roughness height or deposition velocity, may be required for a given model. The elucidation of these other parameters may not be trivial.

Furthermore, if the model was created for unobstructed flat terrain, but the real terrain is not flat, inaccuracies will result unless the model is altered to suit the real situation. A meteorologist or other mathematical modeler is required for making such alterations.

Another modeling issue is the source geometry. Some models are better than others for a particular source geometry. A model which treats point and volume sources well might not be as good for area sources, for example. Furthermore, the use of a point source approximation for an area source will cause an underestimate of emissions for a measured downwind concentration. The closer the downwind receptor is to the area source, the greater will be the error. A rule of thumb sometimes used by the EPA for square area sources is that the receptor must be a minimum of ten site lengths from the source for the point source approximation to be reasonable.

Some information on dispersion models is available on an EPA computer bulletin board called TTN (Technology Transfer Network). The number to call for modem connections is 919-541-5742. Upon reaching the main menu, choose the "SCRAM" (Support Center for Regulatory Air Models) option for model information.

If one does use a model which accounts for deposition, the model will typically require the sizing of the particles emitted from the dust source. This is because particles of different aerodynamic diameter will deposit on the ground between the source and the sampler at different rates. To model the deposition rate of the dust requires knowledge of the size distribution of the dust. This has often been obtained aerodynamically with cascade impactors, but may also be obtained using other methods.

Exposure profiling method

The exposure profiling method was developed by Midwest Research Institute, under an EPA contract, as a tool for deriving emission factors (Cowherd et al, 1974). The exposure profiler consists of a number of ambient samplers (typically four or five) at several heights along a vertical tower, typically four to ten meters in height (Figure 1). The samplers are provided with a means to sample nearly isokinetically: typically this consists of either interchangeable nozzles of various sizes or variable flow-rate control. Wind speed is monitored by anemometers, usually at two to five heights along the tower (McCain et al, 1985). Wind speeds for unmonitored heights are often calculated using a logarithmic algorithm (Muleski et al, 1993; Axetell and Cowherd, 1984). Wind direction is monitored by a wind vane.

One or more towers of this type is placed downwind of the source, with the sampler intakes pointed into the wind. The profiling tower is placed close to the source, often approximately five meters away (Muleski et al, 1993; Cowherd and Kinsey, 1986; Cuscino et al, 1983;). Ambient samplers (typically between one and four of them) are placed upwind of the source at one or more heights (Pyle and McCain, 1985). The upwind samplers are also placed close to the source, often ten to fifteen meters away (Muleski et al, 1993; Cowherd and Kinsey, 1986; Cuscino et al, 1983). Sampling at the upwind samplers is not necessarily isokinetic (Bohn, 1982).

Exposure (Garman and Muleski, 1993a) may be defined as the net passage of mass through a unit area perpendicular to the plume transport direction (wind direction):

$$E = (10^{-7})CUt$$

where: E = dust exposure (mg/cm^2)
C = net concentration (ug/m^3)
U = approaching wind speed (m/s)
t = sampling duration (s)

Values of exposure will vary at different sites within the plume. The integral of exposure evaluated over the cross section of the plume should equal the total mass flux of dust emitted from the source (Garman and Muleski, 1993a; Axetell and Cowherd, 1984; Bohn et al, 1978). The integration may be accomplished via Simpson's rule. Simpson's rule necessitates an odd number of data points at equal intervals; if additional data points are required to obtain an odd number or equal spacing, they are obtained by extrapolation (Muleski et al, 1993).

In the case of a uniformly emitting "line" source (really a "point" source moving along a line), such as a car moving along a relatively uniform dirt road, a single vertical integration should be sufficient to characterize the emissions (Bohn et al, 1978). In the case of "point" or small area sources, a two dimensional integration will be required (Garman and Muleski, 1993a; Bohn et al, 1978).

Similarly, in the case of a point source moving along a line and emitting uniformly, one profiling tower may suffice to characterize the plume. In the case of "point" or small area sources, a number of profiling towers must be used.

The samplers should be symmetrically placed in the body of the dust plume so that approximately 90 percent of the mass flux of the dust cloud passes between the outermost edges of the array.

As an example, for a Gaussian dust plume, the exposure values measured by the samplers at the edge of the sampling array should be about 25 percent of those measured at the center of the array (Bohn et al, 1978).

Exposure profiling has been used primarily for measuring emissions from sources whose plumes will not have significant mass passing above the highest sampler on a profiling tower. This has largely constrained this method to sampling close to the source. Axetell and Cowherd (1984) for example, write that it is preferable for the profiling towers to be approximately five meters from the source. However, Clayton et al (1984) report the use of sectional aluminum masts to raise the heights of their highest samplers well above 20 meters. This kind of tower height would permit sampling farther from the source. Sampling farther from a point or area source, however, also requires a more horizontally widespread tower array, because of horizontal plume dispersion.

The exposure profiling method may not be practical for sampling large area sources. The bigger the distance between the upwind side of the area source and the profiling tower, the higher the tower will need to be. The longer the dimension of the area source perpendicular to the wind, the wider the profiling array must be.

Exposure profiling uses a mass conservation approach (Garman and Muleski, 1993a) to calculate emission rates from mass fluxes measured downwind. But some PM-10 may deposit on the ground between the source and the profiling tower. This "lost mass" of PM-10 could be significant, particularly if the source is close to the ground. Any deposition occurring between the source and the profiling tower will lead to inaccuracies (under-predictions) in calculating emission rates. The significance of these inaccuracies is unknown.

However, perhaps a distinction should be drawn between the actual emission rate and the relevant emission rate. What we are normally concerned about is entry of dust into the ambient environment. The dust that is immediately deposited is not usually of great concern. Hence, it may be reasonable to acknowledge this source of inaccuracy in the exposure profiling method in terms of measuring the actual emission rate, while realizing that this inaccuracy may not pertain to the "relevant" emission rate.

This inaccuracy could become problematic if the calculated emission rate is to be used with a dispersion model to predict downwind ambient impact. If a dispersion model with a deposition algorithm is used, there will be under-prediction of the ambient impact. "Lost mass" deposited between the source and the profiler will lead to a lower-than-actual calculated emission rate, and then the deposition algorithm will further decrease the predicted downwind concentration.

Nor would it necessarily be correct to use a dispersion model without a deposition algorithm to calculate the ambient impact of a source. Again, in this case, missing mass deposited between the source and the profiler will lead to underestimates of the actual emission rate. The application of a dispersion model without a deposition term tends to lead to overestimates of PM-10 downwind impacts. The result of combining an underestimated emission rate with an ambient impact overestimation is unclear. Possibly the errors would essentially cancel. Perhaps comparing the resulting ambient impact predictions with predictions derived from receptor models provides a clue, but receptor models for dust generally have their own problems with conservation of mass issues.

In any case, the magnitude of the mass lost to deposition between the source and the profiler is unknown. It will vary with source height, meteorological conditions and source-profiler distance. This mass may not be significant at many emission heights and under certain meteorological conditions, but it could be important for sources emitting close to the ground. This mass should be quantified. We would then be more sure of actual emission rates.

Exposure profiling has another source of inaccuracy in the necessity of extrapolating mass fluxes from the outermost samplers in the array to the fluxes outside of the array. The more widespread the sampling array, the more this source of error can be minimized. As an example of the potential magnitude of this source of error, Muleski et al (1983) found between a ten and seventeen percent discrepancy from using a six-meter profiling tower compared to their results using a ten-meter tower, for measuring dust emissions five meters from an unpaved road.

Exposure profiling is considered significantly more accurate than the upwind-downwind method (Kolnsberg, 1982; Fitzpatrick, 1987). This is because exposure profiling samples quasi-isokinetically, typically samples a much larger portion of the dust plume, and does not depend on dispersion modeling for determining emission rates. Kolnsberg (1982) writes that the accuracy of the exposure profiling method is comparable to that of the roof monitor method.

The report of Axetell and Cowherd (1984), which has been included as Appendix F, contains a description of the exposure profiling method and step by step calculations for measuring emission rates from line-sources. Garman and Muleski (1993b) has a less detailed but more current plan for measuring line-source emission rates; this is Appendix H. Another report by Garman and Muleski (1993a) includes information on the calculation of emission rates from area sources, sampling configuration diagrams, and information on sample handling and analysis, and is included as Appendix I.

Portable wind tunnel method

The portable wind tunnel was used in the 1970's to study the effects of wind-blown sand on vegetation, and to quantify the determinants of wind erosion (Fryrear, 1971; Gillette, 1978). It has since been used to quantify wind-generated emissions from exposed soil and from coal storage piles (Axetell and Cowherd, 1984; Cowherd, 1983; Cuscino et al, 1983). It should be reiterated that this method is used only to quantify wind-generated emissions.

The portable wind tunnel is diagrammed in Figure 2 (from Cuscino et al, 1983). The "working" part of the wind tunnel has an open floor and is placed directly on the surface to be tested. An airtight seal is maintained between the tunnel sides and the tested surface (Axetell and Cowherd, 1984). A fan draws air through the tunnel from an intake "upwind" of the test area. At a threshold speed, dust will be picked up or eroded by the passing air stream. The quantity of eroded material (neglecting deposition) is the net amount of dust leaving the tunnel, or the total amount leaving minus the amount entering.

As shown in Figure 2, the emissions sampling in the portable wind tunnel is done in a raised, fully enclosed duct, downstream from the working section. In the past, emissions have been measured isokinetically by ambient sampling equipment. The Emissions Measurement Branch of EPA prefers the use of standard stack sampling trains whenever feasible. This would mean using Method 201 or 201A. An ambient sampler could, however, be used to obtain the concentration of dust in the ambient intake air for the tunnel.

The emission rate calculation is like a stack problem: The emission rate equals the net particle concentration times the tunnel flow rate divided by the cross-sectional area of the "working" part of the tunnel. The calculation of the tunnel flow rate is complicated, however, by boundary layer considerations, including shear stress at the tunnel floor and walls. Axetell and Cowherd (1984) present a calculation procedure for determining flow rate (See pages 82-86 of Appendix F).

Cowherd (1983) stated that the wind speed profile near the tunnel floor followed a logarithmic pattern and was related to friction velocity, roughness height and the distance from the tunnel floor. Friction velocity is related to shear stress at the tunnel sides and floor (White, 1986). Roughness height has been obtained via an extrapolation of the measured wind speed profile; the distance from the tunnel floor at which the wind speed extrapolates to zero is considered to be the mean roughness height (Axetell and Cowherd, 1984). According to Cowherd (1983), knowing the roughness height allows the use of the tunnel centerline wind speed to extrapolate the probable wind speed at 10 meters height via a logarithmic wind profile which describes wind speeds in the atmospheric boundary layer. In practice, this extrapolation is done graphically plotting height versus wind speed using semi-log paper (Cowherd, C., personal communication, 1993). The measured wind speeds are extrapolated "back" to the y-axis to obtain the roughness height, and they are extrapolated "forward" to 10 meters to obtain the wind speed at that altitude. The slope of the graph will be the friction velocity.

Thus over flat ground, the tunnel centerline wind speed can be related to a corresponding wind speed at 10 meters altitude. Since the tunnel centerline wind speed can also be related to a PM-10 emission rate, the wind speed at 10 meters can be related to that emission rate.

For storage piles, the procedure is as above, except that one must also consult EPA publication AP-42, section 11.2.7 in order to obtain the relationship between the unobstructed atmospheric wind speed profile and the wind speed profile at various sites across a storage pile. Section 11.2.7 of AP-42 is included as Appendix J. For a description of the use of the portable wind tunnel see Appendix F (Axetell and Cowherd, 1984).

A basic assumption made in using the portable wind tunnel method concerns the relating of emission rates in the tunnel to those out of the tunnel. Consider a wind speed measured in the open air at a height of 15 cm. That wind moving over a particular segment of open ground at a certain time causes a specific emission rate. Now consider the same wind speed measured at the same height, but moving through a tunnel placed next to the same spot at the same time. It is assumed that if the ground is uniform, the emissions will be the same in and out of the tunnel. In other words, the physical presence of the tunnel is assumed not to affect the emission rate.

The portable wind tunnel method, like the exposure profile method employs a mass conservation approach (Axetell and Cowherd, 1984). Therefore any deposition which occurs between the point of emission and the point of measurement will lead to an underestimation of total emissions. However, one must ask whether such deposition is relevant. Are we concerned with the total flux of PM-10 up from a source, regardless of whether some of it is deposited before it leaves the source, or are we concerned with the net flux leaving the source and entering the ambient environment?

Let us look at the situation in which a dispersion model is used to determine downwind ambient impact of the source. If the source is treated as a point source in a dispersion model with a deposition algorithm, the deposition occurring in the tunnel might not be relevant. This is because the source is actually an area, but is being treated as a point. Deposition occurring within the area of the source but unaccounted for in the tunnel may be accounted for by the deposition algorithm of the dispersion model. (However, one must make sure to consider ambient impact far enough downwind so that the use of a point source model for an area source will not distort the predicted downwind impact--one must be far enough downwind so that the source "looks like" a point.)

Wind erosion of soil or other materials is a complicated process. For example, Cowherd (1982) has suggested that wind gusts rather than mean wind speed cause most particle uptake. Another complication is that wind erosion is not a steady state process, but changes as a function of the amount of erodible material exposed to the wind, which itself is partly a function of the length of time a surface has been exposed to a particular wind speed. The amount of erodible material will also depend upon the frequency, extent, timing and effect of disturbances caused by outside forces acting on a surface to be tested. An example of such outside forces might be the driving of a vehicle on a material storage pile. Cowherd (1983) has dealt with the issue of erosion potential and describes a means to quantify it (Also see Appendix F, pages 85-86). The issue of disturbance will presumably need to be dealt with by having a sampling strategy which fairly represents the normal conditions of the surface to be tested.

However, there are other complications of wind erosion. For example, fetch is defined as the length of exposed surface along the axis of the wind. Gillette (1978) found that increasing the fetch in the portable wind tunnel increased the emission rate per unit area for particles smaller than 25 um. This finding held for all fetches tested, the largest of which was 21.7 cm.

Axetell and Cowherd (1984) use a fetch of 3.5 meters; perhaps this longer fetch obviates this problem, but this is not addressed in the emission measurement literature.

A possibly related issue is that of sandblasting, which is defined as the impaction of saltating particles onto a surface. On open stretches of bare ground, sandblasting causes emissions of particles smaller than 25 um (Gillette, 1978). But in the wind tunnel, Gillette found that emission of particles smaller than 25 um was independent of sandblasting. He speculated that this might be due to the short fetch of the test section in his tunnel. Again it is possible that a 3.5 meter fetch would obviate this problem, but this does not appear to be addressed in the literature on emission measurement. On the other hand, most fugitive dust sources have shorter fetches than those encountered by Gillette on the farmlands of Kansas and Texas. Perhaps sand blasting is unimportant for short fetches.

Gillette (1978) also found during field studies that for some soil types, the ratio of fine to coarse particles emitted increased with increasing wind speed. He wasn't able to duplicate this finding in his wind tunnel. He speculated that this was due to the small fetch of his tunnel inhibiting sandblasting effects.

As a benefit of working primarily in rather flat, unforested areas, both Cowherd and Gillette were able to use values of roughness height extrapolated from measured wind tunnel velocities alone. But this could be a problem in forested or rolling areas where a different means of obtaining roughness height may be necessary (Cowherd, C., personal communication, 1993).

In any case, it appears that the portable wind tunnel is superior to other methods of quantifying wind erosion. Nearly the entire plume is captured. Sampling is isokinetic. Flow rate through the tunnel can be accurately determined.

Scale model wind tunnel method

The scale model wind tunnel method involves the construction of a reduced-size re-creation of a process or landscape inside of a wind tunnel. An attempt is usually made to make important parameters in the wind tunnel resemble those occurring in the field. These parameters may include turbulence, wind shear, or other physical quantities.

Specific approaches to ensuring similarity between the wind tunnel environment and the field environment have differed. There does not appear to be a consensus on the correct approach to take.

Visser (1992) studied the effects of moisture and wind speed on the dust emission rates of three different types of coal. He differentiated emissions occurring from windsift (particles entrained by wind out of a falling stream) from those occurring by impaction (falling and "bouncing"). He determined impaction emissions (dustiness) using a technique described by Lundgren (1986). By dumping the coal into a grille-covered box recessed in the tunnel floor, Visser claimed to minimize re-entrainment of impaction emissions when he was studying windsift.

Emissions were measured isokinetically at nine points downstream from the falling coal. Emission rates were determined by considering the flux at each sampler as representative of the flux of the surrounding area, calculating the flux for each area and then summing the fluxes. The calculated emission factors did not agree well with those from cited field studies, although they were said to be in rough agreement with those from a cited wind tunnel study.

Visser seems to have made the assumption that phenomena observed in his wind tunnel will be indicative of those occurring in the real world. He does not appear to have used any kind of non-dimensional similarity analysis, of the kind often used in scale model wind tunnel studies, even though he was dumping much smaller quantities of coal than would be dumped in real industrial situations. Not only is the different throughput of coal at issue, but the turbulence inside the tunnel is also important. Does the tunnel turbulence at a given wind speed resemble that encountered in real situations? Does the velocity profile in the tunnel resemble that of the atmospheric boundary layer? Visser does not seem to have addressed these issues.

De Faveri et al (1990) studied the effects of wind breaks and coating compounds on emissions from coal storage piles. They built a scale model terrain. In the building of their model, they considered the simulation of the atmospheric boundary layer, the simulation of atmospheric turbulence, and the simulation of terrain with the appropriate roughness height. In relating tunnel design to real-world characteristics, their dimensionless analysis considered the threshold speed (speed at which eroding particles become airborne), air speed, particle size, space, and time of exposure. Interestingly, they scaled the particle size of the coal they were using. The actual measurement of emissions was only quantitative relative to baseline emissions, however.

No method for measuring the actual mass flux was used.

Yocom et al (1985) dropped sulfur into a hopper in a wind tunnel to study dust emissions at wind speeds up to eight miles per hour. In considering the similarity between the atmosphere and the wind tunnel, they explain that the calculation of the Reynolds number for wind tunnels is related to the dimensions of obstructions in the tunnel. They use the square root of the frontal area of a wind flow obstruction as the characteristic length for calculation of the Reynolds number. Wind tunnel turbulence was compared to atmospheric turbulence via a comparison of Reynolds numbers; it was admitted that particularly at low wind speeds, the wind tunnel might not accurately represent atmospheric turbulence.

Another feature of the Yocom study was isokinetic sampling at the downwind end of the tunnel using hi-vol samplers with directional nozzles and variable flow rate. Deposition in the tunnel was measured by weighing deposits on removable aluminum plates placed on the tunnel floor downwind of the dropped sulfur.

An emission factor developed in the Yocom et al study agreed closely with one developed in the field by another group using exposure profiling to measure emissions from the dropping of sulfur. Interestingly, in the Yocom et al study, particles deposited downwind of the dropped sulfur were not included in the calculation of the emission factor, so the actual mass flux out of the stream of dropping sulfur must have been underestimated.

Billman and Arya (1985) studied the effects of windbreaks on wind speeds across downwind storage piles. While they did not directly study emissions, their report is interesting in that a subsequent field study (Zimmer et al, 1986) was performed to verify the results obtained by Billman and Arya. For piles unscreened by windbreaks, Zimmer et al found that while the measured field wind speeds agreed well with those predicted from the wind tunnel studies for measurements taken at the front of storage piles, there was poor agreement at the back of the piles. For the case in which the pile was screened by a windbreak, only one test was directly comparable between the two studies; in that case, the wind tunnel values for screen efficiency were approximately forty percent higher than the field results. Zimmer et al attributed at least part of the discrepancy between field and wind tunnel results to higher turbulence in the atmosphere than in the wind tunnel.

Williams (1982) made the assumption that turbulence in his wind tunnel resembled that at the outdoor site he was modeling. He did not do any non-dimensional similarity analysis. His study is interesting, however, in that he weighed removable dust trays to determine mass flux. He claimed to differentiate between flux occurring by saltation and that occurring by suspension. To do this he used a method involving three adjacent dust trays arranged sequentially along the axis of the wind and embedded in the wind tunnel floor. He claimed that the saltation process reaches equilibrium "quickly." Since the upwind tray receives no saltating particles from other trays, the weight loss measured will be due both to suspension of particles into the air and to any outgoing saltation which occurs. By contrast, the downwind tray should, Williams claims, experience incoming saltation flux from the middle tray equal to that lost downwind to the tunnel, and so net saltation flux of the downwind tray should be zero. Any loss of tray weight in the downwind tray should be due, according to Williams, to suspension alone. It may be, however, that the downwind tray is also incurring deposition of suspended particles eroded from the upwind trays. This would complicate Williams' scheme.

Viner et al (1982) point out that a large wind tunnel cross section is desirable so that boundary layer effects of the walls and ceiling of the tunnel will not complicate the velocity profile around the model. However, a large cross section requires a large fan if high wind speeds are desired.

The Viner study used roughness elements in the tunnel floor to simulate the atmospheric boundary layer. Viner et al state that "The most important parameter with regard to particle entrainment is the shear stress at the surface of the dust sample." Given the roughness elements used in their tunnel, they calculated that the shear stress in the tunnel was typical of atmospheric conditions.

Viner et al note that an advantage of scale model wind tunnel tests is that individual parameters affecting dust emissions can be controlled. A disadvantage is that the relationship between the tests and actual field emissions is "uncertain at best."

The Viner study used three methods for studying emission rates. The information in the published report on the first two methods is limited; however, one method measured mass flux by means of a probe and the other method used a probe to collect particles for optical sizing. The third method was judged the most direct and reproducible. This consisted of weighing a removable tray containing the erodible material, before and after a test. This technique was criticized as being subject, however, to error from the handling of the tray.

Tracer method

The tracer method uses either a gas or particles as a tracer for dust. Several gas tracer studies have used sulfur hexafluoride as a tracer. Usually particulate tracers are fluorescent or phosphorescent or have a dye or other coating which makes them fluoresce or phosphoresce.

The assumption behind the tracer method is that the dispersion of dust will be imitated by the tracer. In other words, the tracer plume will strongly resemble the dust plume if the tracer is released in the same place at the same time as the dust. The validity of this assumption will be discussed later. However, if we assume for the moment that this assumption is correct, then the dust emission rate may be easily determined (Vanderborght et al, 1982):

$$C_d/C_t = Q_d/Q_t$$

where C_d = downwind net dust concentration
C_t = downwind net tracer concentration
Q_d = dust emission rate
Q_t = tracer emission rate

The concentrations of dust and tracer are measured at the same locations upwind and downwind of the source. The upwind concentrations of dust and tracer are subtracted from the respective downwind concentrations to obtain C_d and C_t. (In practice the upwind tracer concentration will be close to zero.) The tracer emission rate is known. (In the case of a gaseous tracer, the gas cylinder can be weighed before and after the tracer release.) Consequently, the emission rate of the dust will be the only unknown quantity and can be readily calculated using the simple proportion expressed above.

Baxter (1983) used sulfur hexafluoride as a tracer for dust from a mining operation. As previously mentioned, an assumption made in this and other tracer studies is that if the tracer is released in the same area and at the same time as the dust, then the tracer and the dust will disperse in similar ways. Another assumption made in this particular study is that deposition of particles less than 30 um in diameter will be minimal over distances less than 100 meters. This latter assumption was necessary because Baxter was measuring gaseous tracer and total suspended particulate at distances as far as 100 meters downwind, and any particulate deposition in that distance would mean that the tracer and the dust were dispersing differently, since sulfur hexafluoride does not undergo deposition.

The assumptions of similar dispersion and no particulate deposition are questionable; their veracity should depend upon emission height and meteorological conditions. For example, if the emissions are close to the ground, significant dust deposition might occur over 100 meters, especially under certain weather conditions. Also, significant reflection of the sulfur hexafluoride gas from the ground could occur over 100 meters. By contrast, the dust would not be expected to undergo much reflection since most dust tends to stick where it impacts.

Baxter visually determined the sites of maximum dust emissions and placed the sulfur hexafluoride cylinders in those areas. He outlined a means of keeping the release rate of the tracer gas constant using a two stage pressure regulator, a fine metering valve and a rotameter. The total amount of gas released was determined by weighing the gas cylinder before and after the tracer gas release.

Baxter used a continuous sulfur hexafluoride analyzer and ambient samplers, all mounted on a van approximately 75 meters downwind of the source. He used the measurements made by the continuous sulfur hexafluoride analyzer to indicate where to move the mobile platform so that he could follow the wind shifts and remain in the main part of the dust plume. Time-integrated samples of sulfur hexafluoride were also obtained using bag samplers.

Vanderborght et al (1982) point out the advantages of using sulfur hexafluoride as a tracer: it is inert, non-toxic, stable up to approximately 500 degrees Celsius, easily detectable at concentrations as low as 50 nanograms per cubic meter, and normal background levels are below the level of detection. Their study used sulfur hexafluoride as a tracer for antimony (Sb) dust emitted from an Sb metallurgical plant.

The Vanderborght study used bag samples of sulfur hexafluoride and used gas chromatography to analyze the samples. Ambient samples of Sb were obtained, and were analyzed using neutron activation and x-ray fluorescence.

Vanderborght et al sampled at distances as close as 15 meters and as far as 180 meters from the source. They make the claim that at these distances deposition of Sb aerosol is negligible. They do admit to problems with the tracer study at the close in distances, however. An indication of such problems is that they found different ratios of C_d/C_t at various sampling sites close to the source. But this ratio should be constant over a given time period, even at different locations, since that ratio should equal Q_d/Q_t and the latter ratio will average to a constant over the same time period. Vanderborght et al attributed this problem to poor mixing of the dust and tracer plumes. This is quite plausible since they were using one point source of sulfur hexafluoride to approximate two separated point sources of dust.

Nevertheless, they found that further downwind, the C_d/C_t ratio remained constant ("within acceptable limits") at various distances and locations. This is evidence both that deposition is negligible at the sampling distances downwind, and that the dust plume and tracer plume disperse in essentially the same way.

Wachter (1980) developed emission factors for stone crushing operations using sulfur tetrafluoride as a tracer gas. He used a gas chromatograph with an electron capture detector to analyze the gas samples.

Wachter made major errors in his paper. Although he was interested in total suspended particulate rather than PM-10, his errors are instructive. First, in arguing for the validity of the tracer technique, he makes the unsupported assumption that particles under 50 um in diameter behave in the same way that sulfur tetrafluoride does. Then, in an effort to prove that only small particles emit past the plant boundaries, he attempts to show, using Stokes's Law, that particles larger than 19 um will settle within 300 meters from the source under average meteorological conditions. Now if particles from 19 um to 50 um in diameter settled within 300 meters from the source, they would certainly not be acting like a gas, and the tracer study would probably be invalid.

Furthermore, the use of Stokes Law alone to determine where atmospheric dust will settle is erroneous. Wachter assumes that the terminal settling velocity along with a horizontal wind speed can be used to calculate where particles will deposit. His approach ignores atmospheric turbulence, which is often the most important determinant of where suspended particles will settle. <u>Deposition velocity</u> rather than terminal settling velocity is generally the most important quantity in such a situation.

Reynolds (1980) was concerned with the re-entrainment or resuspension into the air of hazardous materials deposited on surfaces. He seeded various surfaces with known amounts of phosphorescing particulate tracer having a size distribution in the 1 um to 5 um diameter range. The tracer particles were composed of "zinc-cadmium sulfide." (The EPA does not recommend the use of cadmium-containing materials as tracers.) Reynolds eroded the labeled surfaces using a hi-vol drawing through a portable wind tunnel, and trapped the eroded particles on a filter. Mass loading of the tracer on the filter was obtained using optical techniques. However, since only the mass of tracer was obtained, and not the mass of eroded dust, C_d could not be obtained. So Q_d could not be directly calculated.

Thus, Reynolds was obliged to determine the mass flux of the dust indirectly. He did this by determining a tracer resuspension rate (fraction of tracer particles resuspended in the air per unit time) with a dimension of $time^{-1}$. He notes that initial resuspension fluxes are directly proportional to the resuspension rate, and that "Therefore resuspension fluxes and relationships should be nearly equivalent to functional relationships determined for the resuspension rate...". He then calculates the mass flux of dust based upon estimates of the amount of erodible material available and the calculated resuspension rate for the tracer. He claims that his resuspension rates are accurate to within a factor of three based upon estimations of the magnitudes of the sources of error in the experiment.

The portable wind tunnel method seems to be a much more direct and efficient means of measuring wind erosion than the particulate tracer method described by Reynolds. The mass of eroded dust may be directly calculated with a portable wind tunnel; there is no need to use a tracer as a surrogate for dust. Sehmel (1973) used zinc sulfide particles as a tracer material in a study on dust emission from a paved road. The zinc sulfide was placed on one lane of the road. An array of non-isokinetic samplers was mounted on towers at various distances downwind of the road. Deposition samplers were also positioned at various downwind distances. A graphical integration of the downwind tracer exposure and ground deposition was performed to calculate the resuspension rate per vehicle pass. The quantity of erodible material per unit area of road must be estimated to permit the calculation of the mass flux of dust from the resuspension rate of tracer. The emission rates thus calculated were said to be accurate within a factor of three, based upon an error analysis. The exposure profiling method has often been used to calculate dust emissions from roads in the years since Sehmel's study. Exposure profiling appears to be a superior method in that the dust mass flux is measured directly, rather than using a tracer as a dust surrogate.

The use of gaseous tracers, however, appears promising, particularly for PM-10, the dispersion of which should be more like a gas than the dispersion of total suspended particulate would be (since PM-10 will undergo less deposition). However, the distance at which downwind deposition of PM-10 ceases to be negligible remains to be shown. At the distance where deposition ceases to be negligible, the gas and the dust plumes will be acting differently, and the tracer method will be less valid. This distance will vary with source height and with meteorological conditions, and could be predicted using dispersion models.

By contrast, there is also a problem very close to the source: How do we know that the dust and the tracer have adequately mixed and have formed a uniform plume? Perhaps this issue can be minimized by carefully selecting dust source geometry and tracer source location to facilitate plume mixing. Maybe the problem can be solved by sampling both dust and tracer at a number of locations and distances. If the C_d/C_t ratio is constant over a number of locations and distances, perhaps we can assume, as Vanderborght et al suggested, that this is adequate evidence of plume homogeneity over those areas.

Balloon method

Balloon sampling is an offshoot of the exposure profiling method. The balloon sampling method consists of ambient samplers sampling quasi-isokinetically, suspended at a number of heights from a balloon. Mass flux is computed in the same way as in the exposure profiling method. The balloon method has been used in attempts to sample large area sources or sources which may not be closely approached. Armstrong and Drehmel (1982) designed one such system. Axetell and Cowherd (1984) used balloon sampling in an attempt at measuring the dust emissions from blasting operations.

The latter study had problems with sampling often being non-isokinetic, as well as encountering a problem of being unable to sample a sufficiently large segment of the plume except under very limited wind conditions. The problem of anisokinesis occurred because nozzles on the ambient sampler intakes could not be changed with the balloons aloft, and the flow rate to the samplers was fixed. In this particular instance, variable flow rate to the samplers might have been a good method of maintaining isokinetic sampling. However, isokinetic sampling is less critical for accurate measurement of PM-10 than it is for total suspended particulate (Davies, 1968). Appendix F has a detailed description of the balloon sampling protocol used by Axetell and Cowherd.

Error, accuracy and precision in the methods

Error may be defined as "the departure of the measured value from the true value" (Taylor, 1990). It is equivalent to the term "inaccuracy."

Rosbury et al (1984) focus on error in emission factors. However, some of the sources of error which they mention are broadly applicable to several measurement methods. They place error sources into five categories: emissions, activity parameters, source location, meteorological inputs and dispersion model.

A potentially relevant error that Rosbury et al list in the emissions category is any assumption made about particle size distributions. An example is the common assumption that various types of dust are log-normally distributed.

Errors in defining activity parameters, while not causing inaccuracy in the mass flux measurement itself, can create error in interpreting the meaning of the measurement. Is a given level of activity (which relates to a given mass flux measurement) peak, average or below average activity?

An example of a source location uncertainty may be observed in trying to define source height. For instance, what is the source height for the dust emitted by vehicle traffic on a road?

Uncertainties in meteorological inputs include errors in measurements of wind speed and wind direction. Additional uncertainty comes from estimation of stability class and mixing height. Also, how uniform are the meteorological conditions over the source-measurement area?

Some uncertainties implicit in the use of dispersion models were discussed in the upwind-downwind section of this report. Rosbury et al used three different emission factors in all combinations with three different dispersion models (while holding other variables constant) and thus calculated nine different predicted downwind concentrations. They found that while the emission factors differed by as much as a factor of 4.7, the predicted downwind concentrations differed by as much as an order of magnitude.

Axetell and Cowherd (1984) performed an error analysis on the exposure profiling method and on the upwind-downwind method (See pages 45-46 and Table 3-6 in Appendix F). An error analysis is an attempt to quantify inaccuracy by listing each perceived source of error, deciding whether it is random or systematic, and making an estimate of its potential magnitude and direction. Their initial results indicated that error in the exposure profiling method for particles less than 15 um ranged from -14 percent to +8 percent. Field experience caused them to revise this estimate to plus or minus 30-35 percent. An initial error analysis for the upwind-downwind method estimated inaccuracies of plus or minus 30.5 percent and 50.1 percent for line sources and point/area sources respectively.

Sehmel (1973) and Reynolds (1980) performed error analyses on the different particulate tracer technique each was using, and each claimed that the technique he was using was accurate to within a factor of three.

Error analyses may be useful, but they are essentially an educated guess at the amount of inaccuracy in a method. Even if the estimates of magnitude of known sources of error are good, there is no guarantee that one has considered all sources of inaccuracy. For example, the error analysis of Axetell and Cowherd (1984) for exposure profiling does not appear to take into account the mass balance deficit from deposition that probably occurs with that method.

Turning specifically to the issue of accuracy, this may be defined as the closeness of a method's measurements to the actual value of the measured quantity (Taylor, 1990). To ascertain the level of accuracy of a measurement method, we must know the actual value of the quantity that is being measured.

There may be only one example in the accessible literature in which experimental releases of known quantities of fugitive dust were measured in order to determine the accuracy of a method. Hu Gengxin et al (1992) found that their dispersion model used with the upwind-downwind method predicted emissions within a factor of two of measured emissions, 80 percent of the time. They apparently measured emissions with the quasi-stack method as a reference. However, their experimental technique is not described in detail in their paper, no doubt due to space constraints, so their exact procedure, and consequently its validity, is not entirely certain.

While the quasi-stack method may be, from general principles, potentially the most accurate fugitive dust measurement technique, one must demonstrate that the method does not alter the emissions of dust from the source. This may not be a straightforward task. Consequently, the use of the quasi-stack method as a reference method for determining emission rates appears questionable.

However, an adaptation of the quasi-stack method as a means for determining the accuracy of other methods might work very well. In this case, it would only be necessary that the mass flux of the dust emitting out of the quasi-stack duct equal the mass flux measured by the sampling train inside the duct. In other words, one would need to ascertain that there was negligible deposition in the duct downstream of the sampling train. Then one would have a known emission rate with which to assess the accuracy of other methods.

There appears to be at least one other study using known emission rates of dust to determine the accuracy of dust measurement methods. Hu Gengxin et al cite a book by Li Zhuongkai (1985), presumably written in Chinese, which is said to report on field experiments verifying diffusion models using known releases of glass beads and fog droplets from point sources.

Because so little work has been done comparing known emission rates of dust with measurements made by fugitive dust measuring methods, there is not much to say about the accuracy of these methods, other than what one can deduce or conjecture from general principles. For example, we might expect that methods which sample a large part of a dust plume will be more accurate, on average, than those which sample a small part of the plume. Another generalization is that isokinetic sampling is better than non-isokinetic sampling, although the importance of this decreases as particle size decreases. Dispersion modeling introduces a source of error.

One or more of these generalities might be difficult to quantify. In any case, that would be a tangential approach to defining accuracy. Much more work needs to be done using known emission rates to evaluate the accuracy of fugitive dust measurement methods.

Similarly, few studies have evaluated the precision of methods. Precision may be defined by considering a series of measurements of a particular quantity. The closer the values of the measurements are to each other, the more precise the measurement method (Taylor,1990).

Precision may be a difficult parameter to obtain for fugitive dust measurement methods. This is because it is necessary to have multiple measurements of the same quantity to obtain precision. But it may not be easy to emit the same quantity of dust multiple times. So the papers which report values for precision are those which use methods which obtain multiple measurements of the emission rate during each time period when dust is emitted. These methods are the upwind-downwind method and the tracer method.

Carnes et al (1982) found, in five test runs of the upwind-downwind method, that the coefficients of variation of emission rates (the sample standard deviation divided by the sample mean for each test) ranged from 0.219 to 0.456. There were twelve to fifteen observations in each of the five test runs. Each observation stems from one downwind concentration measurement taken from each ambient sampler in each test run. Carnes et al found that these observations were normally distributed when they were all grouped together.

Vanderborght (1982), using a gaseous tracer, found relative standard deviations (coefficients of variation multiplied by 100%) of 19, 22, 23 and 33 percent in four test runs. Each test run consisted of seven tracer measurements taken more than fifteen meters downwind of the source.

A number of papers submit emission factors to statistical scrutiny. However, one cannot easily obtain the precision of the measurement method from the emission factor statistics because the emission factors are relationships between emission rates and activity levels (such as the number of grams of dust emitted per kilogram of coal handled). Uncertainty in the relationship between the mass flux measurement and the activity level, as well as uncertainty in measurements of the activity itself would complicate any attempt to obtain precision of the measurement method from statistics about the emission factor.

Conclusions

The quasi-stack method may potentially be very accurate, and is probably the best method for measuring emissions from enclosable sources, but difficulties arise in trying to demonstrate that the enclosure of a source does not alter its emissions. Many hood configurations exist which might work with this method, but most have not been studied in the context of measurement of mass flux.

The roof monitor method is probably the best method for measuring emissions from buildings. Sampling problems may include difficulties in adequately sampling very large openings, as well as very variable flow through the openings.

The upwind-downwind method may be the least accurate but most generally applicable of the well established methods. The use of dispersion modeling involved with this method is a major source of error; the dispersion model to be used should be carefully chosen and applied to minimize this source of error.

The exposure profiling method seems to be the best method for unenclosable sources which are of relatively small area and which are amenable to having profilers placed within a few meters of them. The method does have a potentially significant mass balance deficit due to deposition; this deficit should be quantified or at least modeled (using a dispersion model, for example).

The portable wind tunnel method is probably the best method for determining rates of wind erosion. This method also has a potentially significant mass balance deficit which should be quantified or modeled.

A number of more or less experimental techniques have been used. Balloon sampling has encountered some difficulties outside of very specific meteorological conditions. The scale model wind tunnel method has been used in a number of experiments, but differing protocols, non-dimensional analyses, and measuring techniques have been used from study to study. The use of the tracer method has been reported in several papers; while particulate tracers do not appear to have been especially accurate, the gas tracer technique seems promising.

Very little work has been done comparing known emission rates with the measurement of those rates. Consequently, almost no conclusions of a quantitative or definitive nature can be drawn about the accuracy of the measurement methods for fugitive dust. Few studies have been done on the precision of the methods. Much work remains to be done in these areas.

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