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Sampler Placement to Determine Emission Factors from Ground Level Area Sources

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Abstract. *Emission factors are used by state air pollution regulatory agencies to regulate emissions from sources of atmospheric pollution. Emission factors for ground level area sources (GLASs) can be determined by sampling ambient pollutant levels and using dispersion modeling to back-calculate the emission rate from a source. This manuscript describes a method used to locate samplers downwind of a GLAS to most accurately determine the emission flux from the source. Contributions from other sources, changes in particle size distributions, and measurement uncertainty are considered.*

Keywords. Emission factor, area source, Gaussian modeling, particulate matter, PM

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Introduction

Particulate matter (PM) emissions from ground level area sources (GLASs) are important for several reasons. First, the Environmental Protection Agency (EPA) has granted authority to state air pollution regulatory agencies (SAPRAs) to regulate PM emissions from major and minor sources through the permitting process. A facility is considered a major source if it has the potential to emit (PTE) more than 100 tons per year (in an attainment area) of any criteria pollutant, including PM_{10}, from a point source (USEPA, 40CFR Part 70). Minor sources are those sources not meeting the threshold values for emissions. PM_{10} is the portion of particulate in the air having an aerodynamic equivalent diameter (AED) less than or equal to 10µm. Emissions that cannot reasonably pass through a stack, chimney, vent, or other functionally equivalent opening are considered fugitive emissions (USEPA, 40CFR Part 70). A facility's PTE is determined using industry specific emission factors. If the PTE of a source exceeds the threshold for major source classification, a Title V permit is required for that source. The Title V permit establishes the annual fees that a facility must pay to the SAPRA based on the emissions inventory of all emitted criteria pollutants including fugitive emissions. While most agricultural operations are classified as minor sources for PM_{10}, emissions of other criteria pollutants such as ozone and carbon monoxide in non-attainment areas may require agricultural operations to obtain Title V permits.

Additionally, PM_{10} emissions from GLASs are of interest due to a special use of the National Ambient Air Quality Standards (NAAQS) as property line concentrations not to be exceeded. The NAAQS were established in 1977 to protect public health and welfare by limiting the permissible ambient concentrations of criteria pollutants below established levels. For PM_{10}, the current NAAQS (established in 1987) is 150 $\mu g/m^3$ (24-hour average). While the NAAQS implementation guidelines specify that samplers must be placed in areas where they are not affected primarily by any one source (Watson et al, 1997), some SAPRAs have begun using the NAAQS as property line concentrations not to be exceeded. Property line concentrations can be determined by using emission factors to model downwind pollutant concentrations or by placing samplers at or beyond the property line of a pollutant source. However, due to the high labor and capital requirements of conducting a sampling campaign, dispersion modeling using established, industry specific emission factors is most often used to demonstrate compliance with this special use of the NAAQS.

In order to determine PM emission factors from GLASs, field sampling must be conducted and used in conjunction with dispersion modeling to determine area fluxes for PM. A protocol for determining fugitive emissions from GLASs using dispersion modeling and measured downwind concentrations is described by Wanjura et al (2004). For the emission factors developed using this protocol to be accurate, samplers must be located such that they are affected primarily by the source of interest, the uncertainty associated with modeling the source is minimized, and the particle size distribution (PSD) of the collected sample is representative of the PSD of the emitted PM.

The following paper describes a method used to locate samplers downwind of a GLAS to most accurately determine the emission flux from the source. Because of the prevalence of Gaussian dispersion models on the EPA's list of approved dispersion models (USEPA, 40CFR Part 51) (only one of the six approved models is not Gaussian based) and because Gaussian modeling is the most commonly used technique for predicting the impact of non-reactive pollutants (Trinity Consultants, 2000), the Gaussian method of dispersion modeling will be analyzed here. However, similar techniques may be applied to Lagrangian Stochastic models as well. An example 1000m x 1000m area source is used to demonstrate the procedure.

Methods

Pollutant Flux

A complete protocol for determining fugitive emissions from GLASs is described in Wanjura et al (2004). The process is briefly described here to lay sufficient groundwork for discussion of sampler placement.

The first step in determining emission fluxes from a GLAS is to collect measured downwind pollutant concentrations and meteorological data. This data is then used in a dispersion modeling program to back-calculate an emission flux (Q_2) according to equation 1:

$$\left(\frac{C_1}{Q_1}\right) = \left(\frac{C_2}{Q_2}\right) \quad (1)$$

where: C_1 = modeled downwind concentration (μg/m^3) using an initial flux Q_1 (μg/m^2/s),

C_2 = measured downwind concentration (μg/m^3), and

Q_2 = source emission flux (μg/m^2/s).

To back-calculate the emission flux from a source that would result in a particular measured downwind concentration, a pollutant dispersion model must be used. For the dispersion model to accurately characterize the flux from a source, the PM deposited on the filter must have originated from the source of interest, or a protocol for characterizing the contribution from other sources (including fluxes and PSDs) must be employed.

Contributing Area

Without known emission fluxes from the area surrounding the source of interest (as is most often the case), it is important to maximize the influence that the area source of interest has on the measured PM concentration. To do this, samplers should be placed downwind of the area of interest. Because wind direction is not constant, the sampler should be placed as close to the source of interest as possible to reduce the effects of shifting winds.

For example, consider a 1000m x 1000m cattle feedyard in an area with a predominantly southern wind. Consider two receptors placed 500m from the western edge of the feedyard - one 10 m from the northern edge and one 200 m from the northern edge (figure 1).

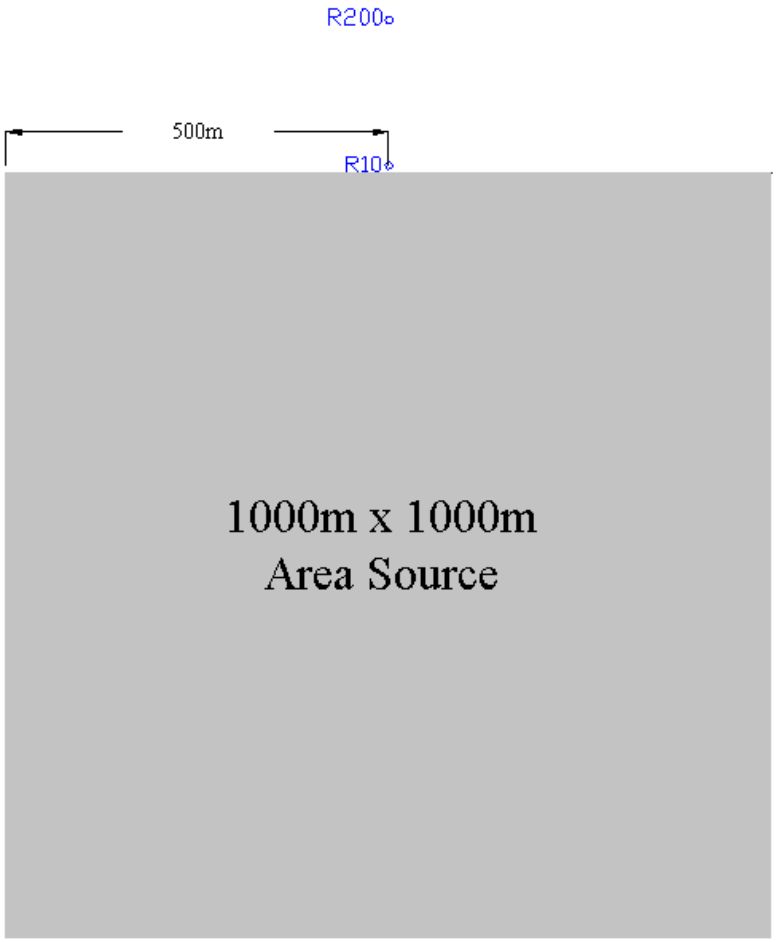

Figure 1. Map of feedyard for example calculations with receptors at 10m and 200m downwind in the predominant wind direction.

Using a line extended along the southern edge of the feedyard as the upwind boundary (figure 2), the percentage of the contributing area that is within the area of interest for wind directions of 0º, 15º, and 45º from the predominant direction is shown in table 1.

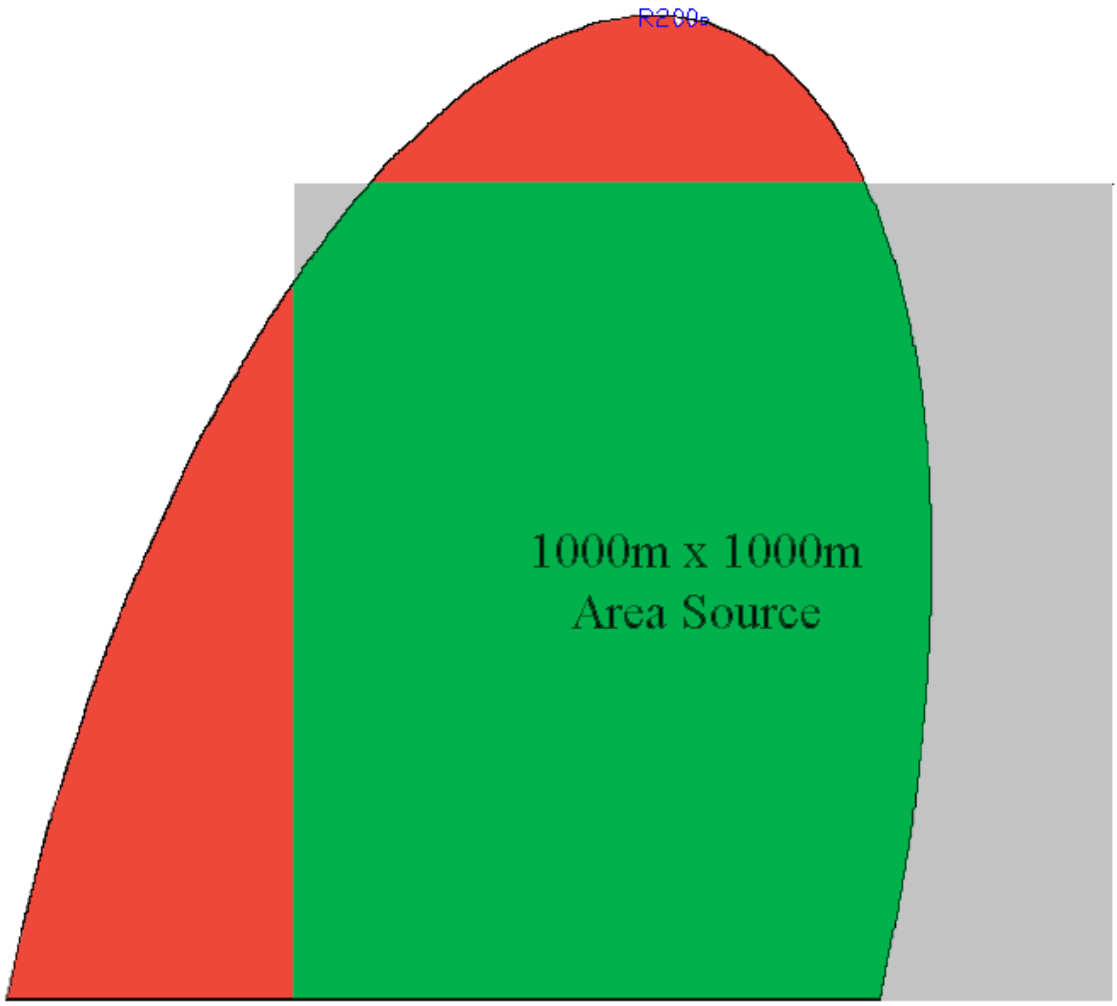

Figure 2. Example of effective area calculation.

Table 1. Percentage of contributing area within source area of interest.

Receptor	Wind Direction		
	0º	15º	45º
R_{10}	99.97%	99.09%	31.47%
R_{200}	95.29%	90.11%	18.58%

As seen in table 1, the percentage of contributing area within the area source of interest is substantially larger for the sampler placed 10m from the source boundary (R_{10}) than for the sampler placed 200m from the source boundary (R_{200}) indicating that the sampler placed closer to the source of interest will give more accurate emission factors than that placed farther downwind. The differences are more pronounced the farther the wind direction is from the line directly between the centroid of the source and the sampler.

Particle Size Distribution

To determine the true concentration of PM_{10}, samples must be collected using total suspended particulate (TSP) sampling inlets rather than PM_{10} inlets, particularly in the presence of PM characterized by a large mass median diameter, as is typical of PM emitted from agricultural industries (Buser et al, 2001). Particle size distribution analyses are then conducted on the PM captured on the filters to determine the fraction of TSP that is PM_{10}. The fraction of TSP that is PM_{10} is then multiplied by the measured TSP concentration to determine the true PM_{10} concentration. In order to determine the true PM_{10} flux from the source, the PSD of PM on the filter must represent the PSD of PM emitted from the source. This can be facilitated by placing the sampler close enough to the source to prevent gravitational settling of large particles.

To determine the optimal sampler location so that the PSD on a TSP sampler filter will be equivalent to the PSD of PM emitted, the effect of gravitational settling on changes in PSD were analyzed using the protocol described by Wang et al (2005). While this protocol does not take into account effects of atmospheric turbulence on particle settling, it does give an idea of how PSD will change as the distance between the source and receptor changes.

As described in Wang et al (2005), the horizontal settling distance was determined according to equation 2:

$$X_{TS} = \frac{h + \Delta h}{V_{TS}} * U \qquad (2)$$

where: X_{TS} = particle horizontal settling distance (m),

h = physical stack height (m),

Δh = plume rise (m),

V_{TS} = particle terminal settling velocity (m/s) determined by Stoke's Law, and

U = wind speed (m/s).

Because the source in this case is a GLAS, a stack height of 0m was used. Based on previous showing that a box model with a 4m box height at the edge of a feedyard yielded comparable results to ISCST3, a plume rise of 2 m (half the total plume height) was used. Particle density was assumed to be 1.5 g/cm^3, and wind speed was assumed to be 3 m/s.

The settling distance of particles up to 60µm in AED were analyzed. As expected, settling distance decreased as particle diameter increased. However, for 60µm, the settling distance was 3710m, indicating that no significant change in PSD will occur due to settling within 3 km of the source. Therefore, in most cases, concerns over changes in PSD with downwind distance should not dictate sampler locations.

Dispersion Modeling

A pollutant dispersion model must be used to back-calculate the emission flux from the source that would result in a particular measured downwind concentration. Gaussian methods are the most commonly used technique for predicting the dispersion of non-reactive pollutants (Trinity Consultants, 2000). Industrial Source Complex Short Term Version 3 (ISCST3) (the current regulatory model) and AERMOD-PRIME (the dispersion model that began replacing ISCST3 on November 9, 2006) both estimate the emissions from an area source by using multiple finite line sources and performing a numerical integration in the upwind direction by considering the space between the lines (Trinity Consultants, 2000).

According to Cooper and Alley (2002), the finite length line source model can be expressed as:

$$C = \frac{q}{2\pi u \sigma_y \sigma_z}\left\{\exp\left[\frac{-(z-H)^2}{2\sigma_z^2}\right] + \exp\left[\frac{-(z+H)^2}{2\sigma_z^2}\right]\right\}\int_{y1}^{y2}\exp\left(\frac{-y^2}{2\sigma_y^2}\right)dy \qquad (3)$$

where: C = steady-state concentration at a point (μg/m^3),

q = average emission rate (μg/m-s),

u = average wind velocity (m/s),

σ_y, σ_z = horizontal and vertical spread parameters (m),

z = vertical distance from ground level (m),

H = effective stack height (m),

y = horizontal distance from plume centerline (m), and

y_1,y_2 = distance from left and right ends of line source, respectively, to receptor (m) (see figure 3).

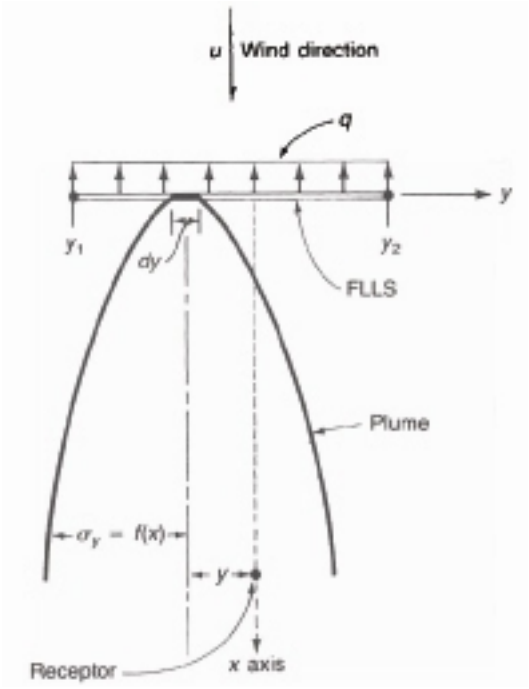

Figure 3. Example of finite length line source with labeled variables (Cooper and Alley, 2002).

For a GLAS, the vertical height (z) and effective stack height (H) are both zero. Equation 3 can also be simplified by defining a variable B such that:

$$B = \frac{y}{\sigma_y} \qquad (4)$$

Substituting equation 4 into equation 3 for a GLAS results in:

$$C = \frac{q}{\pi u \sigma_z} \int_{B_1}^{B_2} \exp\left(-\frac{B^2}{2}\right) dB \quad (5)$$

or

$$C = \frac{2q}{u\sigma_z \sqrt{2\pi}} \left(G_2 - G_1\right) \quad (6)$$

where: G_2,G_1 = the Gaussian distribution function evaluated at B_2 and B_1, respectively.

Uncertainty Estimation

For emission factors to be credible, the uncertainty associated with the calculated (or modeled) flux value must be minimized. Measured downwind concentrations are determined using gravimetric analyses, so the uncertainty in the value of C_2 is a function of the scale or microbalance used to determine the mass of PM accumulated on the filter and the sampler flow rate. Uncertainties in C_2 are not a function of sampler placement.

The use of a dispersion model to determine pollutant flux means that the uncertainty of the calculated flux is a function of the uncertainty of each of the variables used in the model. The uncertainty of the modeled flux can be estimated using the method of uncertainty estimation analyzed by Kline and McClintock (1953). This method, commonly known as the propagation of uncertainty, involves using a first or second order Taylor series approximation to estimate the total uncertainty associated with a measurement. This overall uncertainty results from uncertainty in the measurement of each independent variable propagating through data reduction equations (Coleman and Steele, 1999). The method of uncertainty estimation described here is in accordance with the International Organization for Standardization (ISO) "Guide to the Expression of Uncertainty in Measurement" (ANSI/ASME, 1998).

Assuming that each uncertainty is at the same confidence level (e.g., 95%), let Y be a function of independent variables x_1, x_2, x_3, …, x_n, such that the data reduction equation for determining Y from each x_i is:

$$Y = Y(x_1, x_2, x_3, \ldots, x_n) \quad (7)$$

Then, let ω_i represent the uncertainty of the independent variable x_i, where i ranges between 1 and n. The uncertainty of Y (ω_Y) resulting from the propagation of the uncertainties in each independent variable (x_i) in the data reduction equation can be calculated as the positive square root of the estimated variance, ω_Y^2, from the equation 8 (Holman, 2001):

$$\omega_Y = +\sqrt{\omega_Y^2} \quad (8)$$

The variance (ω_Y^2) is calculated using equation 9:

$$\omega_Y^2 = \left(\frac{\partial Y}{\partial x_1}\omega_1\right)^2 + \left(\frac{\partial Y}{\partial x_2}\omega_2\right)^2 + \ldots + \left(\frac{\partial Y}{\partial x_n}\omega_n\right)^2 \quad (9)$$

The sensitivity coefficient expresses the ratio of the change of the result to a unit change in one input parameter:

$$\theta_i = \frac{\partial Y}{\partial x_i} \quad (10)$$

where θ_i is the sensitivity coefficient.

The variance, then, may be expressed as:

$$\omega_Y^2 = (\theta_1\omega_1)^2 + (\theta_2\omega_2)^2 + ... + (\theta_n\omega_n)^2 \quad (11)$$

The contribution from each uncertainty component to the overall uncertainty of the result is important when trying to determine the primary sources of uncertainty in experimental measurement. The contribution to overall uncertainty of a given measurement is found by dividing the absolute systematic contribution of a given measurement (U_i) by the total absolute systematic uncertainty:

$$\% \text{ Contribution} = \frac{U_i}{\sum_{i=1}^{n} U_i} x100\% \quad (12)$$

The absolute systematic uncertainty contribution (U_i) of a measurement is found using equation 13:

$$U_i = \left(\frac{\omega_i}{2}\theta_i\right)^2 \quad (13)$$

where: U_i = absolute systematic uncertainty contribution of variable i,

ω_i = the uncertainty of variable i, and

θ_i = the sensitivity coefficient for variable i.

Samplers used in determining emission fluxes, then, should be located so as to minimize the total uncertainty associated with the back-calculated flux. When applied to modeling a finite length line source, equation 11 becomes:

$$\omega_Q^2 = \left(\frac{\partial Q}{\partial C}\omega_C\right)^2 + \left(\frac{\partial Q}{\partial u}\omega_u\right)^2 + \left(\frac{\partial Q}{\partial \sigma_z}\omega_{\sigma_z}\right)^2 + \left(\frac{\partial Q}{\partial z}\omega_z\right)^2 + \left(\frac{\partial Q}{\partial G_2}\omega_{G_2}\right)^2 + \left(\frac{\partial Q}{\partial G_1}\omega_{G_1}\right)^2 \quad (14)$$

The final two terms can be eliminated by locating the sampler away from the edge of the source. If the sampler is located more than 3.8 times the value of σ_y from the edge of the source (measured perpendicular to the wind direction), then the values of G_2 and G_1 go to 1 and 0, respectively, and do not change, making the final two terms equal to zero and reducing total variance, thereby reducing uncertainty. The line along which G_2 and G_1 are equal to 1 and 0, respectively, is called the line of maximum concentration (LMC).

Using the feedyard (figure 1) example again, the LMC was determined by predicting concentrations at a line of samplers located 10m and 200m north of the northern boundary for different wind directions. Predicted concentrations within 2% of the maximum predicted concentration were considered to be along the LMC. A longer LMC represents a better opportunity to obtain samples that represent well the emissions of the source by eliminating the influence of changes in σ_y. The lengths of the LMC for wind directions between 0 and 45º from the predominant wind direction at 10m and 200m north of the example feedyard are shown in table 2.

Table 2. Length of LMC at 10m and 200m north of GLAS as a function of wind direction.

Receptor	**Length of LMC (meters)**

Distance (m)	0°	15°	30°	45°
10	750	675	400	125
200	650	550	200	50
LMC_{10}/LMC_{200}	1.15	1.23	2.00	2.50

The length of LMC is greater at 10 m than at 200 m for each case, with the ratio of LMC_{10} / LMC_{200} increasing as the flow vector gets further away from 0°. Therefore, samplers located 10 m from the feedyard will more likely be within the LMC than samplers located 200 m from the feedyard, thus reducing the overall uncertainty of fluxes calculated using concentrations measured along the LMC.

With the exception of the third term in equation 14, the remaining variables (if the sampler is located along an LMC) are dependent on the precision of instrumentation used to collect data. Returning to the feedyard example, we can determine the uncertainty of calculated fluxes from R_{10} and R_{200} assuming constant southerly winds at a velocity (u) of 3 m/s and an atmospheric stability class of C. If the actual flux from the feedyard were 5.66 $\mu g/m^2/s$ (15 #/1000hd-d assuming 150 ft^2/hd), the predicted concentrations at R_{10} and R_{200} using ISCST3 would be 88.054 and 41.506 $\mu g/m^3$, respectively. Assuming the true flux is unknown, and these concentrations were measured, the flux and associated uncertainty can be calculated.

For this analysis, a variable uncertainty for concentration measurements of 12% (typical for gravimetric analysis with a low volume sampler), an anemometer precision of ±0.5 m/s, and a 20% uncertainty in σ_z [according to Cooper and Alley (2002), for a line source σ_z may be 500% greater than published σ_z value for a point source] were assumed. Using ISCST3, concentration measurements from both receptors would return a flux value of 5.66 $\mu g/m^2/s$, but the uncertainty associated with the flux value using concentrations at R_{10} would be 33.4% while the uncertainty associated with the flux value using concentrations at R_{200} would be 1743%. Concentration measurements contribute 17% of the total measurement uncertainty, wind speed variations account for 34%, and uncertainty in σ_z (which is a function of distance from source to sampler) contribute 49% of the total uncertainty. Therefore, samplers should be placed as close to the source as possible to reduce the uncertainty associated with back-calculated fluxes from measured downwind concentrations.

Conclusion

When determining where to place samplers that will be used to calculate the flux of pollutant emissions from a GLAS, several parameters must be considered, including influences from other sources, uncertainty associated with calculations, and changes in PSD. From this investigation, the following conclusions were drawn:

- The closer the sampler is to the source, the less influence outside sources will have, especially when the wind blows from a direction other than directly from source to receptor;
- No significant change in PSD from a given source will occur as a result of gravitational settling within 3km of the source;
- Samplers should be placed along the line of maximum concentration to eliminate edge effects. The farther the sampler is from the source, the more likely edge effect interference will occur; and
- Samplers should be placed as close as possible to the source to reduce uncertainty associated with back-calculated fluxes from measured downwind concentrations. For a

sampler placed directly downwind of a 1000m x 1000m area source emitting 5.66 $\mu g/m^2/s$ in a 3 m/s wind, uncertainty in calculated fluxes increased from 33.4% to 1743%.

While PM emissions were the primary focus of this paper, similar techniques may be applied to locate downwind samplers for gaseous emissions as well. In the case of gaseous pollutants, deposition, reactivity, and buoyancy considerations must be accounted for rather than gravitational settling. However, the basic techniques, including the uncertainty analysis, would remain unaltered.

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