# ERRORS ASSOCIATED WITH PARTICULATE MATTER MEASUREMENTS ON RURAL SOURCES: APPROPRIATE BASIS FOR REGULATING COTTON GINS

A Dissertation

by

MICHAEL DEAN BUSER

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

May 2004

Major Subject: Biological and Agricultural Engineering

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# ABSTRACT

Errors Associated with Particulate Matter Measurements on Rural
Sources: Appropriate Basis for Regulating Cotton Gins. (May 2004)
Michael Dean Buser,
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Agricultural operations across the United States are encountering difficulties complying with current air pollution regulations for particulate matter (PM). PM is currently regulated in terms of particle diameters less than or equal to a nominal 10 μm ($PM_{10}$); however, current legislation is underway to regulate PM with diameters less than or equal to a nominal 2.5 μm ($PM_{2.5}$). The goals of this research were to determine the biases and uncertainties associated with current $PM_{10}$ and $PM_{2.5}$ sampling methods and to determine the extent to which these errors may impact the determination of cotton gin emission factors.

Ideally, PM samplers would produce an accurate measure of the pollutant indicator; for instance, a $PM_{10}$ sampler would produce an accurate measure of PM less than or equal to 10 μm. However, samplers are not perfect and errors are introduced because of the established tolerances associated with sampler performance characteristics and the interaction of particle size and sampler performance characteristics. Results of this research indicated that a source emitting PM characterized by a mass median diameter (MMD) of 20 μm and a geometric standard deviation (GSD) of 1.5 could be forced to comply with a 3.2 and 14 times more stringent regulation of $PM_{10}$ and $PM_{2.5}$, respectively, than a source emitting PM characterized by a MMD of 10 μm and a GSD of 1.5. These estimates are based on both sources emitting the same concentrations of true

PM or concentrations corresponding to the particle diameters less than the size of interest.

Various methods were used to estimate the true $PM_{10}$ and $PM_{2.5}$ emission factors associated with cotton gin exhausts and the extent to which the sampler errors impacted the PM regulation. Results from this research indicated that current cotton gin emission factors could be over-estimated by about 40%. This over-estimation is a consequence of the relatively large PM associated with cotton gin exhausts. These PM sampling errors are contributing to the misappropriation of source emissions in State Implementation Plans, essentially forcing Air Pollution Regulatory Agencies to require additional controls on sources that may be incorrectly classified has high emitters.

# ACKNOWLEDGMENTS

At times, I believe my family is more excited than myself at the completion of this Ph.D. program. This is not to say that I am not ecstatic about this accomplishment, just an indication of the love and support that I have received from my family. My lovely wife, Susie, my son, G.W., and my daughter, Kaleigh, have been extremely supportive in this endeavor. They have endured not only the late nights associated with a rigorous Ph.D. program, but have tolerated the enormous miles and many hotel rooms associated with working in Stoneville, Mississippi and going to school in College Station, Texas. During the initial journey into this Ph.D. program, our family made some courageous and sacrificing decisions, such as moving Susie and G.W. to College Station seven months prior to my arrival. During this time Susie was the primary parent and I would make the journey from the Mississippi Delta to the Brazos Valley every ten days to spend a few days of quality time with my family. I realize the strain associated with our decision and I would like to thank Susie, G.W., and Kaleigh for their never-ending support and words of encouragement that will be with me for eternity.

Just this morning, Susie told me how proud she was and this was not the first time she had spoken similar words. Although I may not always acknowledge the words of encouragement, I do hear them and they do truly mean a lot. G.W. always has a way with words. During the roughest stretch of this journey, G.W. asked the notorious question “Why do you need a doctorate degree, when you are not going to cure anyone?” G.W. has provided me a wealth of one liners and has consistently reminded me of what is truly important, my family. Kaleigh arrived in the early stages of this journey and has spent many hours with me in front of the computer. Kaleigh has recently begun asking me, “… are you Dr. Daddy, yet?” To Susie, G.W., and Kaleigh, I love you and thank you from the bottom of my heart for everything you have done in making this journey a success.

I would also like to thank my parents: Gary, Gloria, Dennis, Judy, and Gail for their continued words of encouragement, support, crisis counseling, and many more things too numerous to list. Each and every one of you has made a tremendous impact on my life and I am truly proud to have you in my life. Sixteen years ago and two weeks before the Fall 1988 semester began, I was asked the question, “Well are you going to college or just moving out?” I believe I have now answered that question. Parenting is not an easy task, it never ends, you wonder if your children are truly listening, and you worry if your children will make the right decisions. I had some wonderful teachers and I am beginning to understand that the work and effort are worth it in the end, you never want it to end, your children are really listening even when you wish they were not, and, if you were not worrying about your children, you would be worrying about something else. Family is something that is often taken for granted on the surface, but deep down I know how important my parents are to Susie, G.W., Kaleigh, and myself. Thank you!

I would also like to thank my grandmothers, Molena and Zelma. Both of you have been extremely encouraging and supportive in all aspects of my life. I want both of you to know that I think about you each and every day and I enjoy the time we have spent and will spend together. I love you both and you mean more to me than you could ever imagine.

I have had the great fortune of spending quality time with my grandparents and several of my great-grandparents, most of whom have passed on, but are not forgotten. My grandfathers, Guy and Norwood, impacted my life in many ways. Some words of wisdom that took hold were, “Get your education, it’s one thing that no one can ever take away from you” and “Planning is everything, without a plan you will end up chasing your tail and you will wind up in the same place that you started.” My grandfathers also showed me the value of a hard day’s work.

I would also like to thank all the rest of my family and friends. Although our family is relatively small and seems to be spread out across the country, there is a feeling of closeness that exists because of the words and actions that all of you so easily give. During my family’s journey we have made some wonderful friends. Although the miles have extended the time between visits, your phone calls and words of encouragement are truly appreciated.

Dr. Parnell, what can I say! You are outstanding! Throughout my stay in College Station, we had our ups and downs. At times, I thought I would never see graduation. You have provided me much more than just an education; you have provided me the ability to “argue.” Now I know that you and many others will say that I was pretty good before you became my mentor, but I have to say I am now on a whole new playing field. On a more serious note, Dr. Parnell you are an exceptional professor and more importantly an exceptional person. Many of the college professors in the current university systems could learn something from you. It is clearly obvious that your primary focus is on the students. You have made yourself readily available, pushed students to think on the fly, and so many other things that make your students successful agricultural engineers. You have equipped me with knowledge, confidence, common sense, a network, and a drive to succeed. There are no words to describe the magnitude of my gratitude. Thank you!

I would also like to thank my committee members, Dr. Shaw, Dr. Lacey, and Dr. Fuller, for their support and help. I would also like to thank the faculty and staff of the Biological and Agricultural Engineering Department. This group of folks was a pleasure to work with. When I first started college at Oklahoma State University, I enrolled in electrical engineering. By the third semester, I loathed the department. I ventured over to the Agricultural Engineering Department at Oklahoma State University to check out the department. To my surprise, I was able to visit with virtually all the faculty, without making an appointment, and was impressed by the hospitality, facilities, and student awareness. The Biological and Agricultural Engineering Department at Texas A&M has the same atmosphere and is an exceptional department.

To the “crew,” thanks for all the laughs, help, and most importantly the friendship. What a ride! Brad, Ling, Barry, Amber, John, Craig, Clay, and all the rest it was a blast. By the way, thanks for the help. I remember a time not so long ago when Dr. Parnell and I were having one of our discussions in the lab and I could see everyone

and everything that was going on in the office.  As I recall, some of you missed class because you did not want to get involved in the "discussion."  Anyway, with that aside, each of you has a very promising future and I hope that we will have the opportunity to work together in the future.

I would like to thank the USDA-ARS Cotton Ginning Research Unit in Stoneville, MS and the Cotton Production and Processing Unit in Lubbock, TX for providing the financial support necessary for completing this program.  I would like to personally thank Mr. Stanley Anthony and Dr. Alan Brashears for making this journey a reality.  Finally, I would like to thank all the folks within the USDA-ARS, the cotton industry, and others for their continued interest in seeing that I complete this Ph.D. program.

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# INTRODUCTION

The Federal Clean Air Act (CAA) of 1960, and subsequent amendments, established national goals for air quality and incorporated the use of standards for the control of pollutants in the environment. The 1970, CAA Amendments provided the authority to create the Environmental Protection Agency (EPA) and required the EPA to establish National Ambient Air Quality Standards (NAAQS) (USEPA, 1996a). The NAAQS are composed of primary standards (based on protecting against adverse health effects of listed criteria pollutants among sensitive population groups) and secondary standards (based on protecting public welfare, e.g., impacts on vegetation, crops, ecosystems, visibility, climate, man-made materials). In 1971, EPA promulgated the primary and secondary NAAQS as the maximum concentrations of selected pollutants (criteria pollutants) that, if exceeded, would lead to unacceptable air quality (*Federal Register*, 1971). The NAAQS for particulate matter (PM) was established in 1971, and total suspended particulate (TSP) was defined as the criteria pollutant. The CAA Amendments of 1977 required the EPA to review and revise the ambient air quality standards every five years to ensure that the standards met all criteria based on the latest scientific developments. In 1987, the EPA modified the PM standards by replacing TSP with a new criteria pollutant that accounted for particles with an aerodynamic equivalent diameter (AED) less than or equal to a nominal 10 μm ($PM_{10}$) (*Federal Register*, 1987). On July 16, 1997, the EPA promulgated additional NAAQS for PM. This update incorporated an additional criteria pollutant for the ambient air standards that would account for particles with an AED less than or equal to a nominal 2.5 μm ($PM_{2.5}$) (*Federal Register*, 1997).

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This dissertation follows the style and format of the *Transactions of the ASAE*.

Health risks posed by inhaled particles are influenced by both the penetration and deposition of particles in the various regions of the respiratory tract and the biological responses to these deposited materials. The largest particles are deposited predominantly in the extrathoracic (head) region, with somewhat smaller particles deposited in the tracheobronchial region. Still smaller particles can reach the deepest portion of the lung, the pulmonary region. Risks of adverse health effects associated with the deposition of typical ambient fine and coarse particles in the thoracic region (tracheobronchial and pulmonary deposition) are much greater than those associated with deposition in the extrathoracic region. Further, extrathoracic deposition of typical ambient PM is sufficiently low, so particles depositing only in that region can safely be excluded from the indicator (USEPA, 1996a). Figure 1 shows the American Conference of Governmental Hygienists (ACGIH, 1997) sampling criteria for the inhalable, thoracic, and respirable fraction of PM. Note that virtually no respirable PM (PM that can penetrate into the alveolar region of the human lung) is greater than 10 μm, whereas 50% of the 3.5 μm particles are considered respirable and can reach the alveolar region, as shown in Figure 1.

In 1987, the EPA staff recommended that a $PM_{10}$ standard replace the TSP standard. Based on the literature, it was EPA's intent for the $PM_{10}$ sampler to mimic the thoracic fraction of PM (Hinds, 1982). The original acceptable concentration range proposed by the EPA Administrator was 150 to 250 $\mu g/m^3$ $PM_{10}$ 24-hour average, with no more than one expected exceedance per year (USEPA, 1996a). The Administrator decided to set the final standard at the lower bound of the proposed range. The rationale behind this decision was that this standard would provide a substantial margin of safety below the levels at which there was a scientific consensus that PM caused premature mortality and aggravation of bronchitis, with a primary emphasis on children and the elderly.

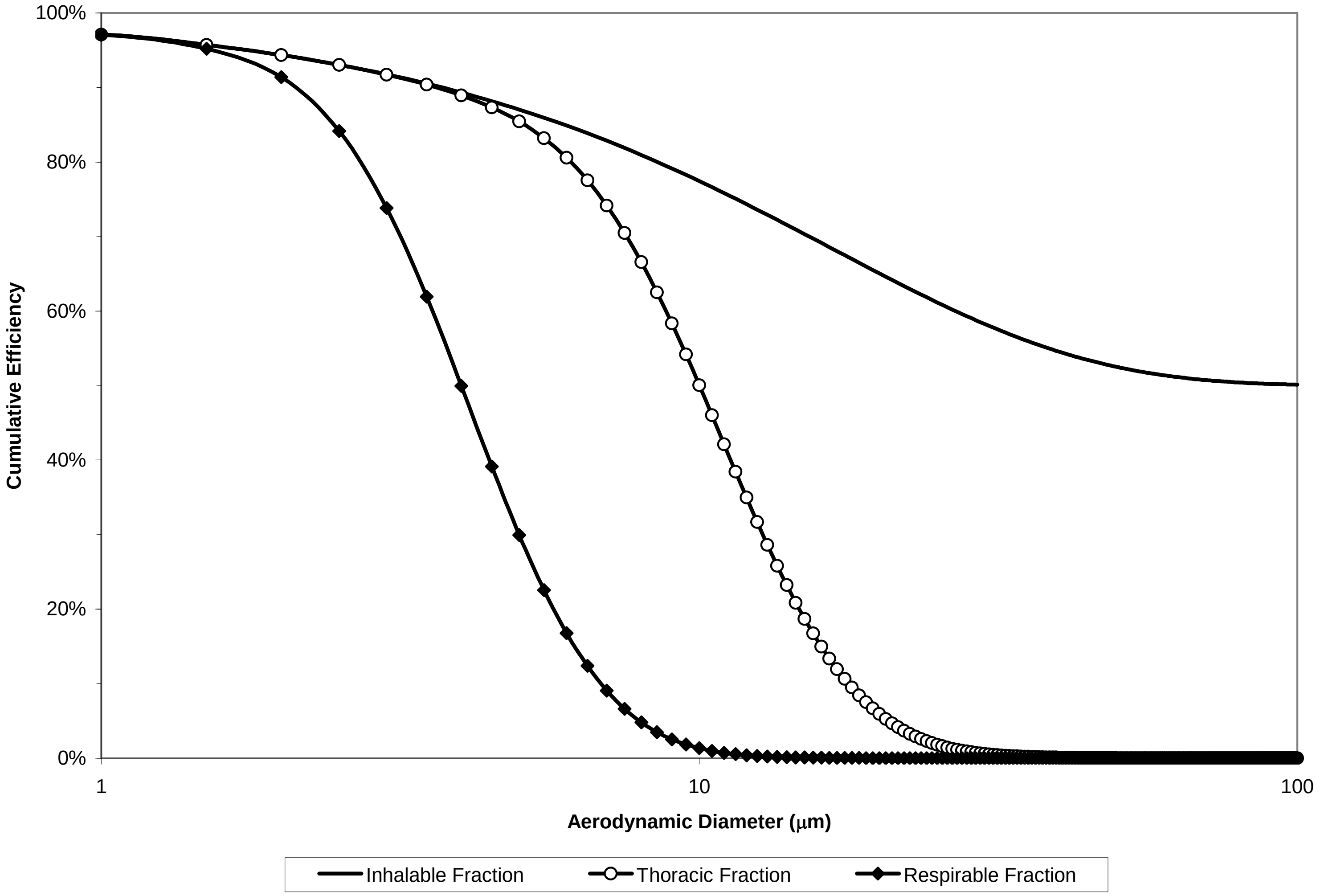


**Figure 1.** American Conference of Governmental Industrial Hygienists sampling criteria for inhalable, thoracic, and respirable fractions of PM (ACGIH, 1997).

In 1979, EPA scientists endorsed the need to measure fine and coarse particles separately (Miller et al., 1979). Fine particles are often associated with the respirable fraction of PM, with typical cut-point values ranging from 3.5 to 5.0 μm for “healthy adults” (ISO, 1993). EPA’s emphasis on the 2.5 μm cut-point was more closely associated with separating the fine and coarse atmospheric aerosol modes, rather than mimicking a respiratory deposition convention. Based on the availability of a dichotomous sampler with a separation size of 2.5 μm, EPA recommended 2.5 μm as the cut-point between fine and coarse particles (USEPA, 1996a). Because of the wide use of this cut-point, the $PM_{2.5}$ fraction is frequently referred to as “fine” particles. It should be noted; however, that ISO (1993) defines a “high risk” respirable convention with a cut-point of 2.4 μm, which is claimed to relate to the deposition of particles in the lungs of children and adults with certain lung diseases.

The NAAQS for $PM_{10}$ and $PM_{2.5}$ are the ambient air concentration limits set by EPA that should not be exceeded (USEPA, 2001a). The regional or area consequences for multiple exceedances of the NAAQS are having an area designated as non-attainment with a corresponding reduction in the permit allowable emission rates for all sources of PM in the area. Some State Air Pollution Regulatory Agencies (SAPRA) are attempting to use the NAAQS as the property line emission limit (standard). For example, if the property line concentration is greater than the NAAQS, the facility in not in compliance. The current $PM_{10}$ primary 24-hour NAAQS is 150 micrograms per actual cubic meter (μg/acm). The secondary NAAQS for $PM_{10}$ is set at the same level as the respective primary NAAQS. The proposed $PM_{2.5}$ primary 24-hour NAAQS is 65 μg/acm. The secondary NAAQS for $PM_{2.5}$ is set at the same level as the primary NAAQS.

Prior to, and since, the inclusion of $PM_{10}$ and $PM_{2.5}$ into the PM regulation numerous journal articles and technical references have discussed epidemiological effects, trends, regulation, and methods of determining $PM_{10}$ and $PM_{2.5}$. A common trend among many of these publications is the use of size-selective samplers to collect information on $PM_{10}$ and $PM_{2.5}$ concentrations.  Size-selective sampler based concentrations are commonly used in comparing $PM_{10}$ and $PM_{2.5}$ emission concentrations from various sources.  All too often, the sampler concentrations are assumed to be accurate measures of $PM_{10}$ and $PM_{2.5}$.  However, issues such as airflow measurement uncertainties, weighing procedure uncertainties, sampler uncertainties, sampler biases, and environmental conditions used in reporting results (dry standard versus actual conditions) will impact the sampler concentration measurements and must be incorporated to obtain accurate $PM_{10}$ and $PM_{2.5}$ concentrations.

The concentration obtained from a PM sampler is only an approximation or estimate of the true concentration and is complete only when accompanied by a quantitative statement of the measurements uncertainty (Taylor and Kuyatt, 1994).  The difference between error and uncertainty is that a measured value can unknowably be very close to the true value, resulting in a negligible error even though the uncertainty associated with the measurement is relatively large.  Accuracy is a qualitative term that corresponds to the degree of agreement between the measured concentration and the true concentration.  Repeatability corresponds to the degree of agreement between the concentrations obtained from successive measurements carried out under the same conditions (e.g. same measurement procedure, same observer, same measuring instrument, same location, and repetition over a short period of time).  Reproducibility corresponds to the degree of agreement between the concentrations obtained under changed measurement conditions (e.g. principle of measurement, method of measurement, observer, measuring instrument, reference standard, location, conditions of use, and time).  Bias or systematic error corresponds to the mean that would result from an infinite number of measurements of the same concentration carried out under repeatability conditions minus the true value.

Agricultural operations are encountering difficulties complying with current air pollution regulations for PM (Cotton Chronicle, 2002).  When air pollution compliance issues arise for a specific facility or operation, air pollution regulatory agencies generally conduct property line sampling or dispersion modeling to determine if the facility or operation is in compliance with the corresponding regulations. Modeling requires emission rates, which are determined either from the EPA list of emission factors (AP-42) or from source sampling. Emission factors are industry specific and are generally based on source sampling studies; however, emission standards are part of a federal guidance and can be impacted by the political process. All property line sampling for compliance purposes generally requires the use of EPA approved samplers. Ideally, these samplers would produce an accurate measure of the pollutant indicator; for instance, a $PM_{10}$ sampler would produce an accurate measure of PM less than or equal to 10 μm AED (true $PM_{10}$). However, samplers are not perfect and errors are introduced because of established tolerances for sampler performance characteristics, interaction of particle size and sampler performance characteristics, and others.

# OBJECTIVES

The goals of this research were to determine the bias and uncertainty associated with the current sampling methods used in regulating PM and to develop procedures so that all industries are equally regulated. This research focuses on the regulation of cotton gins; however, many of the concepts presented will apply directly to other agricultural operations. These operations include, but are not limited to: harvesting, tillage, feedlot, grain elevators, and travel on non-paved roads. Further, the concepts presented may apply to non-agricultural industries when the regulated pollutant is $PM_{10}$, $PM_{2.5}$, and/or $PM_{coarse}$. The specific objectives include:

1) Biases and uncertainties associated with the size-selective pre-separators used in the current and proposed EPA methods of determining $PM_{10}$, $PM_{2.5}$, and $PM_{coarse}$ and calculating the ratio of $PM_{2.5}$ to $PM_{10}$ greatly impact the reported concentrations and will vary based on the emitting source.
2) The particle size distribution characteristics associated with cotton gin exhausts will vary by process stream, and the magnitude of these characteristics will impact the biases and uncertainties associated with current and proposed EPA stack sampling methods resulting in over-estimated cotton gin PM emission factors.

# LITERATURE REVIEW

Air pollution or air quality concerns from the public and/or governing bodies have generally stemmed from single events or air pollution episodes in heavily industrialized societies that resulted in the loss of human life. Examples of such instances were reported by Firket (1931) and Logan (1953). Firket (1931) described the effects of a thick fog that covered the industrial Meuse Valley in Belgium in December of 1930, in which several hundred people were afflicted by sudden acute respiratory symptoms and cardiovascular failure. It was reported that more than sixty people perished after only a few hours of exposure. Firket (1931) estimated that more than 3,000 deaths would have resulted if a similar incident would have occurred in a city the size of London. In December of 1952, such a fog did occur in London. Logan (1953) reported that more than 4,000 deaths were attributed to the four day fog. Similar air pollution episodes have also been reported in the United States. In October of 1948, smog covering the coke and steel producing Monongahela River Valley of Donora, Pennsylvania, resulted in twenty deaths in a city with a population of about 10,000 (USEPA, 1982a). Other United States air pollution episodes have included: the September 1952 incident in Detroit, Michigan and the November 1953 Thanksgiving Day episode in New York City. These types of air pollution events were the driving force behind governments enacting national air quality standards, such as the National Ambient Air Quality Standards in the United States.

The United States National Ambient Air Quality Standards (NAAQS) are promulgated by the United States Environmental Protection Agency (EPA) to meet requirements set forth in Sections 108 and 109 of the United States Clean Air Act (CAA). Section 108 directs the EPA Administrator to list pollutants that may reasonably be anticipated to endanger public health or welfare and to issue air quality criteria for these pollutants. The air quality criteria are to reflect the latest scientific information on the extent of all identifiable effects on public health and welfare expected from the presence of a pollutant in ambient air (USEPA, 2003). Section 109 directs the EPA

Administrator to set and periodically revise, as appropriate, (a) primary NAAQS which in the judgment of the Administrator are requisite to protect public health with an adequate margin of safety, and (b) secondary NAAQS which, in the judgment of the Administrator, are requisite to protect the public welfare from any known or anticipated adverse effects (e.g., impacts on vegetation, crops ecosystems, visibility, climate, and man-made materials). An independent committee of non-EPA experts, the Clean Air Scientific Advisory Committee (CASAC), is to provide the EPA Administrator advice and/or recommendations regarding the scientific soundness and appropriateness of the corresponding criteria and NAAQS. The criteria pollutants currently listed in the NAAQS include: ozone ($O_3$), particulate matter (PM) listed as $PM_{10}$ and $PM_{2.5}$, carbon monoxide (CO), Sulfur Dioxide ($SO_2$), nitrogen oxides ($NO_x$), and lead (Pb). This review will focus specifically on PM related issues.

Particle matter is not a single pollutant, but a mixture of many classes of pollutants that differ in source, formation mechanism, composition, size, and chemical, physical and biological properties. Because PM is not a homogeneous pollutant, measuring and characterizing particles suspended in the atmosphere is a challenging task and there is no perfect method for every application. Particulate matter requires a different interpretation of exposure in contrast to other specific criteria gaseous pollutants, such as CO (Mage, 1985). When a molecule of CO is emitted from a combustion powered vehicle, it is indistinguishable from a molecule of CO emitted from a fireplace; however, a 1 μm aerodynamic equivalent diameter (AED) particle emitted from a combustion powered vehicle and a 1 μm AED particle emitted from a fireplace can have a different shape, mass, chemical composition, and/or toxicity. Since health effects associated with inhalation of PM can depend upon its mass and chemical composition, PM exposure should be measured in terms of mass and chemical composition as a function of the particle size distribution.

Aerosol scientists typically use four different approaches, or conventions, in the classification of particles by size: (1) modes, based on the observed size distributions and formation mechanisms; (2) cutpoint, usually based on the 50% cutpoint of a specific sampling device; (3) dosimetry or occupational health, based on the entrance into various compartments of the respiratory system; and (4) regulatory sizes, used air quality standards (USEPA, 2003). The modal classification, first proposed by Whitby (1978), is frequently approximated by several independent lognormal distributions. Particles in ambient air are usually distributed bimodally in two overlapping size categories. Coarse mode refers to the distribution of particles with diameters mostly greater than the minimum in the particle mass or volume distributions, which generally occurs between 1 and 3 μm (USEPA, 2001a). These particles are usually mechanically generated (e.g. from road construction). Fine mode refers to the distribution of particles with diameters mostly smaller than the minimum in the particle mass or volume distributions, which generally occurs between 1 and 3 μm. These particles are generated from combustion or formed from gases. Particles in these two modal categories tend to differ in terms of formation mechanisms, source of origin, chemical composition, behavior in the atmosphere and human respiratory tract, exposure, dosimetry, toxicology, and epidemiology. Wilson and Suh (1997) suggest that fine and coarse particles are best differentiated by their formation mechanism. Over the years, the terms fine and coarse, as applied to particle sizes, have lost their precise meaning given by Whitby's (1978) definition. Therefore, in any given article, the meaning of fine and coarse, unless otherwise defined, must be inferred from the author's usage. In particular, $PM_{2.5}$ and fine mode particles are not equivalent (USEPA, 1996a).

Particulate material is classified as primary or secondary. Primary PM refers to PM that is in the same chemical form in which it was emitted into the atmosphere; whereas secondary PM corresponds to PM formed in the atmosphere by the transformation of gaseous emissions (USEPA, 1996a). Primary coarse particles are usually formed by mechanical processes but can include sources such as wind-blown dust, sea salt, road dust, and combustion generated particles such as fly ash and soot. Primary fine particles are emitted from sources either directly as particles or vapors that can condense to form ultrafine or nuclei-mode particles (USEPA, 2003). Secondary formation processes can result in either the formation of new particles or the addition of particulate material to preexisting particles. As a result, it is more difficult to relate secondary ambient PM concentrations to sources of precursor emissions than identifying the sources of primary particles (USEPA, 2001a).

Airborne PM can also be classified as anthropogenic or natural in origin. Both anthropogenic and natural PM can occur from primary or secondary processes. Anthropogenic refers to PM that is directly emitted or formed from precursors that are emitted as a result of human activity (San Joaquin Valley Unified Air Pollution Control District, 1996). Primary anthropogenic sources include fossil fuel combustion, fireplace emissions, and road dust. Secondary anthropogenic PM can be generated photo-chemically from anthropogenic $SO_2$, $NO_x$, or organic gases (USEPA, 1996a). Primary natural sources include wind blown dust from undisturbed land, sea salt, and biogenic sources such as pollen, mold spores, leaf waxes, and fragments from plants (Simoneit and Mazurek, 1982). Other biogenic sources include: combustion products of biomass burning caused by lightning; emissions of volatile sulfur compounds from marshes, swamps, or oceans; organic PM formed by the atmosphere reactions of biogenic volatile organic compounds; and particulate nitrates formed by the atmospheric reactions of $NO_x$ emitted from soils (USEPA, 2003). There is an intermediate class of sources associated with agricultural activities which include biomass burning caused by human intervention and the addition of fertilizers to soils resulting in emission of $NH_3$ and $NO_x$. Wildfires have been listed as natural in origin, but land management practices and other human actions affect the occurrence and scope of wildfires. Similarly, prescribed burning is listed as anthropogenic, but can be viewed as a substitute for wildfires that would have otherwise occurred eventually on the same land.

Anthropogenic sources can be further divided into stationary and mobile sources. Stationary sources include fuel combustion for electrical utilities and industrial processes; residential space heating; construction and demolition; wood products processing; mills and elevators used in agriculture; erosion from tilled lands; waste disposal and recycling; and fugitive dust from paved and unpaved roads (USEPA, 2001a, 2003). Mobile, transportation related, sources include direct emissions of primary PM and secondary PM precursors from highway and off-highway vehicles and non-road sources.

The concentration of primary particles in the atmosphere depends on the emission rate of the PM being emitted, transport and dispersion, and removal rate from the atmosphere. Atmospheric lifetimes of particles vary with the particles AED. Primary and secondary fine particles have relatively long lifetimes in the atmosphere (days to weeks) and travel long distances (hundreds to thousands of km) (USEPA, 1996a). These particles tend to be uniformly distributed over urban areas and larger regions. As a result, these particles are not easily traced back to the individual source. Coarse particles normally have shorter lifetimes (minutes to hours) and generally only travel short distances (<100 km). For large particles, buoyancy forces cannot overcome the force of gravity; therefore, gravitational settling becomes an important factor. Coarse particles tend to be unevenly distributed across urban areas and have more localized effects as compared to fine particles; however, dust storms occasionally cause long range transport of small coarse mode particles (USEPA, 2001a). Receptor or dispersion models are the primary means used to estimate contributions of different source categories to PM concentrations at individual monitoring sites (USEPA, 2003). Dispersion models (i.e. three dimensional chemistry and transport models) are formulated in a prognostic manner (i.e. they attempt to predict species concentrations using a tendency equation that includes terms based on emissions inventories, atmospheric transport, chemical transformations, and deposition).

Wind blown dust has been reported as the largest single source of PM in global emissions inventories, constituting about 50% of the total global primary and secondary PM (IPCC, 1995). United States $PM_{10}$ emissions inventories from 1987 to 1996 for various sources are shown in Table 1. Table 2 lists the 1990 California $PM_{10}$ emissions inventory for fugitive dust by air basin. Fugitive dust emissions arise from paved and unpaved roads, building construction and demolition, storage piles, parking lots, mining operations, feed lots, grain handling, and agricultural tilling in addition to wind erosion. Measurements of soil constituents in ambient samples suggest that fugitive dust emissions are highly variable and may only account for 40% of the average $PM_{10}$ mass concentration, much lower than inventory estimates (USEPA, 1996a). Watson and Chow (1999) examined and estimated that uncertainties in emissions inventories could be as low as 10% for the best characterized source categories, whereas emissions from windblown dust and other fugitive dusts should be regarded as order of magnitude estimates.

The 1999 nationwide $PM_{2.5}$ emissions inventory, broken down by categories in Figure 2, suggests that fugitive dust emissions constitute over 50% of the total $PM_{2.5}$ emissions inventory (USEPA, 2001a). Estimates from wind erosion on natural surfaces was not included in Figure 2 because the source is highly sporadic and typically occurs during periods of high winds, resulting in highly variable emission factor estimates. Although fugitive dust emissions constitute about 50% of the total primary $PM_{2.5}$ inventory, they constitute less than 15% of the source strengths inferred from the receptor modeling studies (USEPA, 2003).

## Health Effects

EPA’s primary PM standards are generally based on health effects studies. Health effects studies typically focus on: 1) increases in premature deaths; 2) respiratory and cardiovascular illness; 3) lung function decrements and symptomatic effects, such as those associated with chronic bronchitis; 4) changes to lung structure and natural defense mechanisms; and 5) increases in lost work days and school absences corresponding to PM exposure (USEPA, 1982a). Health effects studies generally fall into one of three categories: 1) epidemiology, 2) dosimetry, or 3) toxicology.

**Table 1.** Summary of the nationwide $PM_{10}$ emissions inventories from 1987 to 1996 (EC/R Incorporated, 1998).

| | Estimated Emissions (million metric tons) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Source | 1987 | 1988 | 1989 | 1990 | 1991 | 1992 | 1993 | 1994 | 1995 | 1996 |
| Fuel Combustion – utility and industrial | 0.45 | 0.45 | 0.45 | 0.54 | 0.45 | 0.45 | 0.45 | 0.45 | 0.54 | 0.54 |
| Fuel Combustion – residential wood and other | 0.73 | 0.82 | 0.82 | 0.54 | 0.64 | 0.64 | 0.54 | 0.54 | 0.54 | 0.54 |
| Metals Processing | 0.18 | 0.18 | 0.18 | 0.18 | 0.27 | 0.27 | 0.18 | 0.18 | 0.18 | 0.18 |
| Other Industrial | 0.73 | 0.73 | 0.73 | 0.73 | 0.64 | 0.64 | 0.64 | 0.64 | 0.64 | 0.64 |
| Open Burning and Other Waste Disposal | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 | 0.27 |
| Motor Vehicles and Off-Highway Engines | 0.82 | 0.91 | 0.91 | 0.82 | 0.82 | 0.91 | 0.91 | 0.91 | 0.82 | 0.82 |
| Wildfires and Managed Burning | 0.91 | 1.54 | 0.82 | 1.09 | 0.82 | 0.73 | 0.73 | 0.91 | 0.73 | 0.73 |
| Agriculture | 6.62 | 6.80 | 6.62 | 4.63 | 4.63 | 4.45 | 4.08 | 4.26 | 4.26 | 4.26 |
| Natural Wind Erosion | 1.45 | 16.42 | 10.98 | 10.98 | 10.98 | 2.00 | 0.45 | 2.00 | 1.00 | 4.81 |
| Paved and Unpaved Roads | 15.06 | 16.60 | 15.97 | 12.25 | 12.34 | 12.07 | 12.61 | 12.61 | 11.61 | 11.52 |
| Construction, Mining, and Quarrying | 11.34 | 10.89 | 10.61 | 4.17 | 3.99 | 4.36 | 4.63 | 5.26 | 3.81 | 4.08 |
| Total | 38.56 | 55.61 | 48.26 | 27.13 | 26.85 | 26.76 | 25.40 | 28.03 | 24.40 | 28.40 |

**Table 2.** Emissions inventory for fugitive dust in metric tons per day (%$PM_{10}$) for California Air Basins in 1990 (Thompson et al., 1991).

| | Air Basin, metric tons per day [TPD] $PM_{10}$ (% $PM_{10}$) | | | | | | |
|---|---|---|---|---|---|---|---|
| Source Category | Lake County | North Central Coast | Sacramento Valley | San Francisco Bay Area | San Joaquin Valley | South central Coast | Total for all 6 Basins |
| Farming Operations | 0.36 | 12.70 | 61.69 | 5.35 | 140.91 | 14.52 | 235.53 |
| | (1.3) | (10.2) | (14.1) | (1.1) | (12.9) | (8.3) | (9.8) |
| Construction and | 1.18 | 9.98 | 51.71 | 74.39 | 50.80 | 30.84 | 218.9 |
| Demolition | (4.3) | (8.0) | (11.8) | (14.7) | (5.2) | (17.6) | (9.7) |
| Entrained Road Dust - | 3.63 | 41.73 | 117.93 | 272.16 | 172.37 | 79.83 | 687.65 |
| Paved | (13.4) | (33.4) | (26.9) | (53.9) | (17.5) | (45.6) | (30.5) |
| Entrained Road Dust - | 6.90 | 27.22 | 87.09 | 33.57 | 136.08 | 18.14 | 309.00 |
| Unpaved | (25.4) | (21.8) | (19.9) | (6.6) | (13.8) | (10.4) | (13.7) |
| Unplanned Fires | 13.61 | 1.27 | 48.99 | 0.73 | 42.64 | 6.44 | 113.68 |
| | (50.2) | (1.0) | (11.2) | (0.1) | (4.3) | (3.7) | (5.0) |
| Natural Sources | 0.09 | 20.87 | 17.24 | 11.79 | 371.95 | 7.89 | 429.83 |
| | (0.3) | (16.7) | (3.9) | (2.3) | (37.8) | (4.5) | (19.1) |
| Other Stationary and | 1.36 | 11.25 | 53.98 | 107.23 | 83.92 | 17.51 | 287.49 |
| Mobile Sources | (5.1) | (8.9) | (12.2) | (21.3) | (8.5) | (9.9) | (12.2) |
| Total | 27.13 | 125.02 | 438.63 | 505.22 | 998.67 | 175.17 | 2,282.08 |
| | (100) | (100) | (100) | (100) | (100) | (100) | (100) |

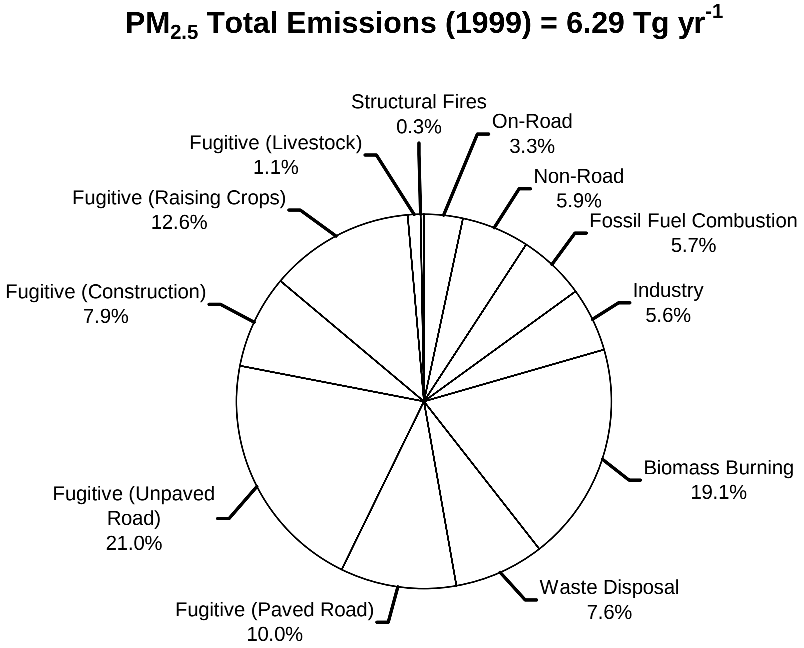


**Figure 2.** Emission of primary $PM_{2.5}$ by various sources in 1999 (USEPA, 2001a).

## Epidemiology

Four of the most common types of epidemiology strategies used in PM health studies, in order of increasing inferential strength are: (1) ecologic; (2) time-series semi-ecologic; (3) longitudinal panel and prospective cohort; and (4) case-control (Rothman and Greenland, 1998). All of these studies are observational rather than experimental based, since participants are not randomly assigned to air pollution exposures. In general, the participant's exposure is not directly observed and the concentration of airborne particles and other air pollutants at one or more stationary air monitors is used as a proxy for individual exposure to ambient air pollution (USEPA, 2001a).

Ecologic study responses are at the community level (i.e. annual mortality rates), as are the exposure indices (i.e. annual average PM concentrations) and covariates (i.e. the percentage of the population greater than 65 years of age) (USEPA, 2001a). No individual data are used in the analysis; therefore, the relationship between health effects and exposure calculated across different communities may not reflect individual-level associations between health outcome and exposure. The use of proxy measures for individual exposure and covariates or effects modifiers may bias the results and within-city or within-unit confounding may be overlooked.

Time series studies are more informative than ecological studies because they allow for the study of associations between changes in outcomes and changes in exposure indicators preceding or simultaneous with the outcome (USEPA, 2001a). Time-series studies use community-level air pollution measurements to index exposure (i.e. daily air pollution concentrations) and community-level response (i.e. the total number of deaths each day by age and/or by cause of death). Prospective cohort studies complement time-series studies; these studies use individual health records, with survival lifetimes or hazard rates adjusted for individual risk factors, and typically evaluate human health impacts of long-term PM exposures indexed by community-level measurements.

A pervasive problem in the analysis of epidemiology data is the unique attribution of the health outcome to the normal causal agent-airborne particles, no matter what design or strategy (USEPA, 2001a). Health outcomes attributed to PM are not very specific and may be attributable to high or low temperature, influenza and other diseases, and/or exposure to gaseous criteria air pollutants. Many of the other factors can be measured directly or by proxies and used as co-variables in the model. However, some of the co-variables are confounders and others are effects modifiers.

Confounding is a confusion of effects. Specifically, the apparent effect of the exposure of interest is distorted because the effect of an extraneous factor is mistaken for or mixed with the actual exposure effect (which may be null) (Rothman and Greenland, 1998). Rothman and Greenland (1998) list three criteria for a confounding factor: (1) a confounding factor must be a risk factor for the health effect; (2) a confounding factor must be associated with the exposure under study in the source population; (3) a confounding factor must not be affected by the exposure of the health effect (i.e. it cannot be an intermediate step in the causal path between the exposure and the disease).

Most extraneous variables fall into the category of effects modifiers (USEPA, 2001a). Effect-measure modification differs from confounding in several ways. The main difference is that confounding is a bias that the investigator hopes to prevent or remove from the effect estimate and effect-measure modification is a property of the effect under study (USEPA, 2001a). In epidemiologic analysis, the goal is to eliminate confounding and estimate effect-measure modification (Rothman and Greenland, 1998). Examples of effect modifiers include environmental variables (e.g. temperature or humidity in time-series studies), individual risk factors (e.g. education, cigarette smoking status, or age in a prospective cohort study), and community factors (e.g. percent of population > 65 years old).

The National Morbidity, Mortality, and Air Pollution Study focused on time-series analyses of $PM_{10}$ effects on mortality during 1987-1994 in the 90 largest U.S. cities (Samet et al., 2000b,c), in the 20 largest U.S. cities in (Dominici et al, 2000), and $PM_{10}$ effects on emergency hospital admission in 14 U.S. cities (Samet et al., 2000,b,c). Results from the Multi-City studies indicated that the percent excess (total, non-accidental) deaths estimated per 50 $\mu g/m^3$ increase in $PM_{10}$ were: 1) 2.3% in the 90 largest U.S. cities (4.5% in the Northeast region); 2) 3.5% in the 8 largest Canadian cities; and 3) 2.0% in western European cities (using $PM_{10}$ = TSP*0.55). These combined estimates are consistent with the range of $PM_{10}$ estimates reported in USEPA (2001a). The $PM_{10}$ relative risk estimates derived from short-term $PM_{10}$ exposure studies reported in USEPA (1996a) suggested that there were 2.5 to 5.0% excess deaths per 50 $\mu g/m^3$ $PM_{10}$ increase. Higher relative risks were indicated for the elderly and for those with pre-existing cardiopulmonary conditions. Schwartz et al., (1996) reported, in their analysis of the Harvard Six City data, that there was a 2.6 to 5.5% excess risk per 25 $\mu g/m^3$ $PM_{2.5}$ increase.

Schwartz et al. (1999) investigated the association of coarse particle concentrations with non-accidental deaths in Spokane, Washington. Coarse particles dominated the $PM_{10}$ data on dust storm days, confirmed by separate measurements of $PM_{10}$ and $PM_1$, in August, 1996. Various sensitivity analyses considering different seasonal adjustments, year effects, and lags, were conducted. Schwartz et al. (1999) concluded that there was no evidence to suggest that coarse (presumably crustal) particles were associated with daily mortality. A previous USEPA (1973) case study reported that most agricultural dusts (crustal material) were not toxic, but could be irritating to the respiratory tract.

In general, results from epidemiology studies indicate that several combustion-related source-types are likely associated with mortality, including: vehicle emissions, coal combustion, oil burning, and vegetative burning (USEPA, 2003). The crustal factor from fine particles was not associated with mortality in the Harvard Six Cities data, and the crustal factor from fine particles in the Phoenix data was negatively associated with mortality. Therefore, the source-oriented evaluations seem to implicate fine particles of anthropogenic origin as being most important contributing factor related to increased mortality and are generally non-supportive of increased mortality risks being related to short-term exposures to crustal materials (USEPA, 2001a).

Some epidemiology studies suggest an association between short-term ambient coarse-fraction ($PM_{coarse}$) exposures (inferred from stationary air monitor measures) and short-term health effects in epidemiology studies (USEPA, 2003). This suggests that $PM_{coarse}$, or some constituent components may contribute to health effects in some locations. Reasons for differences among the various findings reported on $PM_{coarse}$ health effects are still poorly understood, but several of the locations where significant $PM_{coarse}$ effects have been observed (Pheonix, Mexico City, and Santiago) tend to have drier climates and exhibit higher levels of organic particles from biogenic processes (e.g. endotoxins and molds) during warm months. Other studies suggest that $PM_{coarse}$, of crustal origin are unlikely to exert notable health effects under most ambient exposure conditions.

A growing body of epidemiology evidence suggests associations between short- and long-term ambient $PM_{2.5}$ exposures (inferred from stationary air monitor measures) and health effects, concluding that $PM_{2.5}$ (or one or more $PM_{2.5}$ components) is a probable contributing cause of observed PM associated health effects (USEPA, 2003). More recent epidemiology findings suggest that health effects are associated with concentrations of ultrafine (nuclei-mode) particles, but not necessarily more so than for other ambient $PM_{2.5}$ components.

It is likely that differences in observed health effects will be found to depend as much on site-specific differences in chemical and physical composition characteristics of ambient particles as on differences in PM mass concentration (USEPA, 2001a). For example, the Utah Valley study (Dockery et al., 1999; Pope et al., 1991, 1999) showed that $PM_{10}$ exposure while the steel mill was operating (known to be richer in metals) was more highly associated with adverse health effects than $PM_{10}$ exposure while the steel mill was closed.

A large body of epidemiology evidence, implying strong associations between short- and long-term ambient $PM_{10}$ exposure (inferred from stationary air monitor measures) and mortality/morbidity effects, suggests that $PM_{10}$ (or one or more $PM_{10}$ components) is a probable contributory cause of human health effects (USEPA, 1996a). However, there are critical methodological issues associated with these studies including: 1) potential confounding of PM effects by co-pollutants (especially major gaseous pollutants e.g. $O_3$, CO, $NO_2$, $SO_2$); 2) attributing PM effects to specific PM components (e.g. $PM_{10}$, $PM_{coarse}$, $PM_{2.5}$, ultrafines, sulfates, and metals) or source-oriented indictors (e.g. motor vehicle emissions and vegetative burning); 3) temporal relationships between exposure and effect (e.g. lags and mortality displacement); 4) general shape of exposure-response relationships between PM and/or other pollutants and observed health effects (e.g. potential indications of thresholds for PM effects); and 5) the consequences of measurement error (USEPA, 2001a). It is not possible to assign any absolute measure of certainty to conclusions based on the findings of epidemiology studies (USEPA, 2003). Observational epidemiology study findings could be enhanced by supportive findings of causal studies from other scientific disciplines (e.g. dosimetry and toxicology).

Some exposure analysts contend that for community time series epidemiology to yield information on the statistical association of a pollutant with a health response, there must be an association between personal exposure to a pollutant and the ambient concentration of that pollutant because people tend to spend about 90% of their time indoors where they are exposed to both indoor generated and ambient infiltrated PM (Brown and Paxton, 1998; Ebelt et al., 2000). Consequently, numerous epidemiological findings suggest significant associations between ambient PM concentrations and various morbidity and morality health indices in spite of low correlations between ambient PM concentrations and measures of personal exposure, described by some exposure analysts as an exposure paradox (Lachenmyer and Hidy, 2000; Wilson et al., 2000).

Total personal exposure to PM consists of outdoor (ambient) and indoor exposures. Nonambient conditions, mainly indoors at home or at work, occupy the vast majority of a person’s time. A USEPA (1989) report indicates that U.S. residents spend 85.2% of their time indoors, 7.4% in or near a vehicle, and only 7.4% outdoors. $PM_{10}$ in ambient air penetrates into residential microenvironments and reaches an equilibrium approaching outdoor concentrations (USEPA, 1996a). Once indoors, PM of ambient origin decreases because of deposition on surfaces through gravitational settling and electrostatic attraction. Coarse PM has a much higher deposition rate than fine PM. Unless the air exchange rate is very high, the ambient PM that penetrates indoors will be removed by deposition more rapidly than it can be replaced (USEPA, 2001a).

Ambient monitoring stations can be some distance away from individuals and can represent only a fraction of all likely outdoor microenvironments that individuals come in contact with during the course of their daily lives (USEPA, 2003). Furthermore, most individuals are quite mobile and move through multiple microenvironments (e.g. home, school, office, commuting, and shopping) and engage in diverse personal activities at home (e.g. cooking, gardening, cleaning, and smoking). Consequently, exposures of some individuals could be classified incorrectly if only ambient monitoring data was used to estimate total personal PM exposures. Therefore, improper assessment of exposures using data routinely collected by the ambient monitoring stations could lead to increases in epidemiological analysis standard errors.

Between 1982 and 1996, personal and indoor PM exposure studies demonstrated that indoor PM mass concentrations and personal PM exposures were greater than ambient PM mass concentrations when measured simultaneously (e.g. Sexton et al., 1984; Spengler et al, 1985; Clayton et al., 1993). As a result, the NRC (1991) recognized the potential importance of indoor sources of contaminants (including PM) in causing adverse health outcomes. When a cross-sectional analysis was performed, comparing ambient $PM_{10}$ to personal exposures of $PM_{10}$ for a group of subjects, the correlation moved toward zero because of the large influences of indoor sources and sinks that varied between the individuals (USEPA, 1996a).

## Dosimetry

The respiratory tract includes the air passages of the nose, mouth, nasal pharynx, oral pharynx, epiglottis, larynx, trachea, bronchi, bronchioles, and alveoli. Based on the mechanisms associated with deposition and clearance of inhaled aerosols, the respiratory tract can be divided into three functional regions: 1) extrathoracic (ET) or head region; the airways extending from the nasal passages down to the epiglottis and larynx at the entrance to the trachea (the mouth is included in this region during mouth breathing); 2) tracheobronchial (TB) region, the primary airways of the lung from the trachea to the terminal bronchioles; and 3) pulmonary region, the airspaces of the lung, including the respiratory bronchioles, alveolar sacs, atria, and alveoli (i.e., the gas-exchange region) (USEPA, 1982a).

Particles may deposit within the respiratory tract by five mechanisms: 1) inertial impaction, 2) sedimentation, 3) diffusion, 4) electrostatic precipitation, and 5) interception (USEPA, 1982a). Sudden changes in air flow direction and/or velocity cause particles to deviate from the streamlines of airflow, resulting in the particles impacting the airway surfaces (USEPA, 2001a). The ET and upper TB airways are characterized by high air velocities and sharp directional changes and are dominant sites of inertial impaction. Impaction is a significant deposition mechanism for particles larger than 1 μm AED. A particle will acquire a terminal settling velocity when a balance is achieved between the acceleration of gravity acting on the particle and the viscous resistance of the air, resulting in the particles settling out of the air stream and contacting the airway surfaces (USEPA, 1996a). These deposition processes act together in the ET and TB regions, with inertial impaction dominating in the upper airways and gravitational setting becoming increasingly dominant in the lower conducting airways.

The ambient air often contains particles that are too massive to be inhaled. Inhalability is referred to the overall spectrum of particle sizes that are potentially capable of entering the respiratory tract (USEPA, 2001a, 2003). Inhalability is defined as the ratio of the number concentration of particles of a certain AED that enter the nose or mouth to the ambient number concentration of the same diameter particle present in an inhaled volume of air (International Commission of Radiological Protection, 1994). In general, for humans particles greater than 100 μm AED have a low probability of entering the mouth or nose in still air; however, there is no sharp cutoff. As particle AED increases from 1 to 10 μm, nasal region deposition at rest increases from 17% to 71% (NCRP, 1997), allowing more particles in this size range to reach the TB and alveolar regions. Lippmann (1977) calculated that about 10% of particles as large as 15 μm AED might enter the tracheobronchial tree during mouth breathing. The fraction of inhaled particles depositing in the ET region is quite variable and depends on particle size, flow rate, breathing frequency, and whether breathing is through the nose or the mouth. Filtration capabilities associated with mouth breathing are limited in comparison to the nasal airways, resulting in an increased deposition of particles in the lungs (TB and pulmonary regions).

The occupational health community has defined size fractions for use in the protection of human health. This convention classifies particles into inhalable, thoracic, and respirable particles according to upper size cuts (USEPA, 1996a). Inhalable particles enter the respiratory tract, including the head airways. Thoracic particles travel past the larynx and reach the lung airways and the gas-exchange regions of the lung.

Respirable particles are a subset of thoracic particles which are more likely to reach the gas-exchange region of the lung. As of 1993, a unified set of definitions was adopted by ACGIH (1994), ISO (1993), and CEN (1993). The exact shapes of each efficiency curve were mathematically defined by Soderholm (1989) and are slightly different for each convention. Similar thoracic penetration conventions were adopted by ISO (1993), CEN (1993), ACGIH (1994), and USEPA (1987b), each with cutpoint values of 10.0 μm. The EPA definition was based primarily on data from Chan and Lippmann (1980).

The AMA (1963) reported that particles with an AED larger than 10 μm were seldom found in the air spaces of the lungs. Particles larger than 10 μm AED do not pass the filtering mechanisms of the respiratory tract and are of less concern. Recent studies have considered the deposition profiles of particle modes that exist in ambient air in order to provide information on dosimetry particle size fractions. Venkataraman and Kao (1999) examined the contribution of fine and coarse modes of $PM_{10}$ to total lung and regional lung doses resulting from a 24-h exposure concentration of 150 $\mu g/m^3$. The daily mass dose from the $PM_{10}$ exposure for three breathing cycles resulted in 36% of the inhaled coarse PM being deposited in the respiratory tract; 30% in the nasopharynx, 4% in tracheobronchial, and 2% in pulmonary regions. About 9% of the fine particle mass was deposited in the respiratory tract; 1.5% in nasopharynx and tracheobronchial, and 6% in pulmonary regions.

Based on information concerning PM exposure, dosimetry, toxicology, and epidemiology, the overall weight of evidence supports the conclusions that PM, especially fine PM, is the primary contributor to a variety of adverse health effects associated with air pollution (USEPA, 2003). However, technical issues still remain in separating the effects of fine and coarse particles and delineating respective contributions of PM acting along with, or in conjunction with, gaseous co-pollutants in increasing risks of health effects anticipated to occur in response to exposures to contemporary particle-containing ambient air mixes in the United States.

Misra et al. (2002) states, “Although epidemiological studies to date have not made it perfectly clear whether it is particle mass, surface area, or number concentrations that may be responsible for these observed health outcomes presumably attributable to PM, certain toxicological investigations suggest that atmospheric ultrafine particles may be responsible for some of these adverse effects (Oberdorster et al., 1992; Oberdorster et al., 1995; Dreher et al., 1997; Donaldson et al., 1998)". Recent epidemiological studies (Peters et al., 1997) demonstrate a stronger association between health effects and exposures to ultrafine particles as compared to accumulation or coarse particles. Toxicological studies by Donaldson et al. (1998) and Oberdorster et al. (1992) indicated that ultrafine particles exerted a stronger physiological effect than the same mass of coarse or fine particles.

## Regulation

The Federal CAA of 1960, and subsequent amendments, established national goals for air quality and incorporated the use of standards for the control of pollutants in the environment (USEPA, 1996a). The 1970, CAA Amendments provided the authority to create EPA, and required EPA to establish NAAQS. The NAAQS are composed of primary (based on protecting against adverse health effects from listed criteria pollutants among sensitive population groups) and secondary standards (based on protecting public welfare; e.g. impacts on vegetation, crops, ecosystems, visibility, climate, and man-made materials) (Cooper and Alley, 1994). In 1971, EPA promulgated the primary and secondary NAAQS as the maximum concentrations of selected pollutants (criteria pollutants) that, if exceeded, would lead to unacceptable air quality (*Federal Register*, 1971). The CAA Amendments of 1977 required EPA to review and revise the ambient air quality standards every five years to ensure that the standards met all criteria based on the latest scientific developments.

The NAAQS are the concentration limits set by EPA that should not be exceeded (*CFR*, 2001c). The regional or area consequences for multiple exceedances of the NAAQS are having an area designated as nonattainment, with a corresponding reduction in the permit allowable emission rates for all sources of PM in the area. The EPA regulatory authority for PM only extends to the ambient air, defined in 40CFR50.1(e) as that portion of the atmosphere, external to buildings, to which the general public has access. By the operative definition of ambient air, polluted air inside a building, or on property owned or controlled by a private entity, is not regulated by the NAAQS. Myers and Logan (2002) reported that a network of over 1,000 ambient air samplers are currently being used throughout the United States to measure $PM_{2.5}$ and $PM_{10}$ concentrations, and that an additional network of approximately 200 samplers are currently being used to characterize total particulate mass and speciation of the collected particulate.

One method currently being used to determine source specific compliance with air pollution regulations is to measure the public exposure to criteria pollutants and compare to a standard. Some State Air Pollution Regulatory Agencies (SAPRAs) are using the NAAQS as the property line emission limit. If the property line concentration is greater than the NAAQS, the facility is not in compliance. The NAAQS were not intended to be used in evaluating the effectiveness of controls. The standards were originally intended to ensure that ambient concentrations of pollutants were at levels low enough to protect public health (Chow, 1995).

On April 30, 1971, EPA promulgated the original primary and secondary PM NAAQS under Section 109 of the CAA (*Federal Register*, 1971). The reference method for measuring attainment of these standards was the high volume total suspended particulate (TSP) sampler (*CFR*, 1986). The primary TSP standard for PM was set at 260 $\mu g/m^3$, 24-hour average not to be exceeded more than once per year, and 75 $\mu g/m^3$, annual geometric mean. The secondary TSP standard was set at 150 $\mu g/m^3$, 24-hour average not to be exceeded more than once per year (USEPA, 1996a, 2001a, 2003).

On July 1, 1987, EPA published revisions to the NAAQS for PM. The principle revisions in 1987 included: 1) replacing TSP as the indicator for the ambient standards with a new indicator that includes particles with an AED less than or equal to a nominal 10 $\mu m$; 2) replacing the 24-hour primary TSP standard with a 24-hour $PM_{10}$ standard of 150 $\mu g/m^3$, with no more than one expected exceedance per year; 3) replacing the annual primary TSP standard with an annual $PM_{10}$ standard of 50 $\mu g/m^3$, averaged over three years; and 4) replacing the secondary TSP standard with 24-hour and annual $PM_{10}$ standards identical in all respects to the primary standards (*Federal Register*, 1987).

Conceptually, a broad based PM indicator such as TSP set at a stringent level can provide effective protection for the most harmful components. However, because such a standard would set unnecessary controls on extrathoracic constituents unlikely to be harmful, it would not be an efficient standard (USEPA, 1982b). The risks of adverse health effects from extrathoracic deposition of typical ambient PM are sufficiently low that particles depositing only in that region can safely be excluded from the indicator (USEPA, 1996a). Considering these conclusions, other information on air quality composition, the requirement to provide protection for sensitive individuals who may breath by mouth or oronasally, and the thoracic penetration convention adopted by ISO (1981), the EPA staff recommended a size specific indicator that focused on particles with diameters less than or equal to a nominal 10 μm, referred to as $PM_{10}$. Based on the literature, it was EPA's intent that $PM_{10}$ be measured by a sampler with a penetration curve that mimics the thoracic penetration curve associated with the human respiratory system (Miller et al., 1979). With such a cutpoint, larger particles are not entirely excluded, but are collected with a substantially decreasing efficiency and smaller particles are collected with increasing efficiency. Such an indicator ($PM_{10}$) is conservative with respect to health protection in that it includes all of the particles small enough to penetrate to the sensitive pulmonary region and includes approximately the same proportion of the coarse mode fraction that would be expected to reach the tracheobronchial region.

The original $PM_{10}$ concentration range proposed by the EPA Administrator was 150 to 250 $\mu g/m^3$ $PM_{10}$ 24-hour average, with no more than one expected exceedance per year and an annual three year average $PM_{10}$ range of 50 to 65 $\mu g/m^3$ (USEPA, 1996a). The lower bound of this range was derived from the original assessment of the London mortality studies. The upper bound was based on a study conducted by Lawther et al. (1970), which suggested that health effects were likely at PM concentrations above 250 $\mu g/m^3$. Additional evidence suggested that long term degradation in lung function could likely occur at $PM_{10}$ annual levels above 80 to 90 $\mu g/m^3$, with other evidence indicating levels above 60 to 65 $\mu g/m^3$. In light of the 1986 assessment of available scientific data and in accordance with Clean Air Scientific Advisory Committee (CASAC) recommendations, the administrator decided to set the level of the final standards at the lower bound of the ranges originally proposed (i.e., 150 $\mu g/m^3$ 24-hour and 50 $\mu g/m^3$ three year annual average). The rationale behind this decision was that this standard would provide a substantial margin of safety below the levels at which there was a scientific consensus that PM caused premature mortality and aggravation of bronchitis, with a primary emphasis on children and the elderly.

No convincing evidence existed indicating significant adverse soiling and nuisance at TSP levels below 90 to 100 $\mu g/m^3$ and on that basis the Administrator concluded that setting secondary standards different from the primary standards were not requisite to protect the public welfare against soiling and nuisance. This conclusion was supported by CASAC's determination that there was no scientific support for a TSP-based secondary standard. Therefore, the Administrator decided to set 24-hour and annual secondary $PM_{10}$ standards equal to the primary standards in all respects (USEPA, 1996a).

On July 16, 1997, EPA promulgated additional NAAQS for PM and revisions for the existing $PM_{10}$ NAAQS. This update incorporated an additional criteria pollutant that would account for particles with an AED less than or equal to a nominal 2.5 $\mu m$ ($PM_{2.5}$) (*Federal Register*, 1997). EPA's additional $PM_{2.5}$ standards included: 1) 15 $\mu g/m^3$ annual arithmetic mean (averaged over three years) that allowed averaging of multiple community oriented monitors (averaged over 3 years); 2) 65 $\mu g/m^3$ 24-hour average, 98th percentile concentration (averaged over 3 years), maximum population oriented monitor in an area; and 3) the secondary NAAQS for $PM_{2.5}$ were set at the same level as the primary $PM_{2.5}$ NAAQS. The $PM_{10}$ 24-hour standard of 150 $\mu g/m^3$ was retained, but revised to the 99th percentile concentration (three year average).

According to the Federal Register (1997), the pre-1997 $PM_{10}$ standards will remain in effect for an area until the area meets certain criteria. For nonattainment areas that meet the pre-1997 $PM_{10}$ standards, the pre-1997 $PM_{10}$ standards will be revoked when EPA approves the area State Implementation Plans (SIP) that includes all adopted and implemented $PM_{10}$ measures and a section 110 SIP for the revised $PM_{10}$ standard. When nonattainment areas do not meet the pre-1997 $PM_{10}$ standards, EPA promulgates a rule providing for controls that are not less stringent than the controls applicable to areas designated nonattainment before the pre-1997 standards are rescinded, and then the pre-1997 standard will be revoked once the rule is issued. The implementation timeline for the $PM_{2.5}$ standard includes the following deadlines: 1) 1998 through 2000 ambient air samplers will be installed nationwide; 2) 1998 through 2003 EPA will designate nonattainment areas; 3) 2005 through 2008 states must submit implementation plans for meeting the standard; and 4) 2012 through 2017 states have up to 10 years to meet the $PM_{2.5}$ standards (USEPA, 1997).

Following promulgation of the revised PM NAAQS in 1997, legal challenges were filed by several parties addressing a broad range of issues. In May 1998, the U.S. Court of Appeals for the District of Columbia Circuit issued an initial opinion that upheld EPA’s decision to establish fine particle standards, stating that the standards were amply justified by the growing body of empirical evidence demonstrating a relationship between fine particle pollution and adverse health effects (USEPA, 2003). In partial response to numerous challenges to these standards, the U.S. Court of Appeals for the District of Columbia Circuit found in American Trucking Association v. Browner, 175F 3d 1027 (D.C. Cir. 1999) “ample support” for regulating coarse fraction particles, but revoked the revised $PM_{10}$ standards (leaving the 1987 $PM_{10}$ standards in effect) on the basis of $PM_{10}$ being a “poorly matched indicator for coarse particulate pollution” because $PM_{10}$ includes fine particles. Consistent with this specific aspect of the court’s ruling, which EPA did not appeal, EPA is now considering the use of $PM_{coarse}$ as the indicator for coarse fraction particles and the $PM_{2.5}$ standards as the indicator for the fine fraction particles. Therefore, EPA is now developing a Federal Reference Method for the measurement of $PM_{coarse}$.

Another issue concerning the promulgation of the revised PM NAAQS argued before the United States Supreme Court, was that the revised standards were promulgated through an unconstitutional delegation of legislative authority. In February 2001, the U.S. Supreme Court unanimously reversed the Court of Appeals' ruling on the constitutional issue, and sent the case back to the Court of Appeals for resolution of any remaining issues that had not been addressed in that court's earlier rulings (USEPA, 2003). In March 2002, the Court of Appeals rejected all remaining challenges to the standards, finding that the 1997 $PM_{2.5}$ standards were reasonably supported by the record and were not "arbitrary or capricious".

During the development and promulgation of the $PM_{2.5}$ NAAQS, the agricultural community became extremely concerned with the implementation and enforcement of the standards on agricultural related operations. During testimony to Congress's Committee on Agricultural in 1997, Carol Browner, then director of EPA, stated, "EPA does not intend to focus on regulating agricultural tilling to control $PM_{2.5}$ and does not believe it would be efficient for states to do so" (Browner, 1997). Browner (1997) stated that the larger particle size associated with soil particles with relatively low release heights, such as those from tilling operations, will rapidly settle out of the air. Browner (1997) further stated that it is generally believed that almost all $PM_{2.5}$ is secondary $PM_{2.5}$, meaning that it is created by chemical reactions of gasses in the air. Sulfates and nitrates produced by combustion are thought to be the primary gasses responsible for secondary $PM_{2.5}$.

As stated previously, the NAAQS are maximum concentration that should not be exceeded and SAPRA can set lower concentration levels within their jurisdiction. For example, the California Air Resources Board (CARB) has proposed 25 $\mu g/m^3$ 24-hour and 12 $\mu g/m^3$ annual average $PM_{2.5}$ ambient air standards. In addition, CARB is also proposing to lower its $PM_{10}$ annual average standard from 30 $\mu g/m^3$ to 20 $\mu g/m^3$, in comparison to the federal annual average $PM_{10}$ standard of 50 $\mu g/m^3$ (The Cotton Chronicle, 2002).

In 1979, EPA scientists endorsed the need to measure fine and coarse particles separately (Miller et al., 1979). EPA's emphasis on the 2.5 μm cutpoint was more closely associated with separating the fine and coarse atmospheric aerosol modes, rather than mimicking a respiratory deposition convention. Based on the availability of a dichotomous sampler with a separation size of 2.5 μm, EPA recommended 2.5 μm as the cutpoint between fine and coarse particles (USEPA, 1996a). Because of the wide use of this cutpoint, the $PM_{2.5}$ fraction is frequently referred to as "fine" particles. It should be noted that ISO (1993) defines a "high risk" respirable convention with a cutpoint of 2.4 μm, which is claimed to relate to the deposition of particles in the lungs of children and adults with certain lung diseases.

Many observational studies have reported weak, positive associations between rates of mortality in populations and moderate concentrations of $PM_{2.5}$ (Lipfert and Wyzga, 1995). These observational studies have included: cross-sectional studies (Dockery et al., 1993; Pope et al., 1995), in which mortality in various metropolitan areas were associated with ambient concentrations of $PM_{2.5}$; and time-series studies (Samet et al., 2000a), in which daily mortality within a metropolitan area were associated with concurrent or lagged daily fluctuations in ambient $PM_{2.5}$ concentrations. USEPA (1996a, 2001a) and others (Pope, 2000; Ware, 2000) have taken these associations to be causal. EPA has proposed that $PM_{2.5}$ in ambient air be stringently regulated (*Federal Register*, 1997). Although sufficient data on ambient $PM_{2.5}$ have yet to be amassed for portions of the country, indications from many metropolitan areas are that the $PM_{2.5}$ NAAQS will commonly be exceeded (Fitz-Simons et al., 2000), requiring additional controls for emission sources of $PM_{2.5}$. Cost estimates for such controls nationwide range from \$8 to 150 billion annually (Green et al., 2002).

While EPA based its $PM_{2.5}$ standard on epidemiological data that linked mortality with PM concentrations, laboratory studies using controlled human exposure did not produce physiological changes (Cooney, 1998). This uncertainty about the mechanism of action was a key issue in the debate over the final $PM_{2.5}$ standards (Cooney, 1998). Several problematic assumptions were made in crafting the $PM_{2.5}$

NAAQS, including: 1) any and all forms of $PM_{2.5}$ in ambient air cause death with identical toxic potencies; 2) daily and annual, average, mass-based concentrations of total $PM_{2.5}$ are relevant measures for determining public health effects; and 3) decreasing concentrations of ambient $PM_{2.5}$ in any form will decrease rates of death in a reliably quantifiable fashion (Green et al., 2002).

## Particulate Matter Samplers

Particle measurements are needed to determine if a location is in compliance with air quality standards, to determine long-term trends in air quality patterns, and for epidemiologic studies (USEPA, 2003). For these purposes, measurement accuracy is crucial. PM samplers, for the purposes of regulation, fall into one of two categories; ambient or stack samplers. Ambient sampling refers to “the measurement of outdoor air pollutant levels, generally in attempts to characterize fairly broad area pollutant levels” (Wright, 1994). Quantifying pollutant emission rates can be accomplished by source sampling. According to Wright (1994) source sampling is the “measurement of gas flow rate, physical characteristics, composition, and pollutant concentration in exhaust gas streams leaving a process, factory, chimney, or ventilation system and entering the atmosphere”. No size selective sampler is capable of passing 100% of the particles below a certain size and excluding 100% of the particles above that size (USEPA, 1999b). EPA currently defines PM measurement accuracy in terms of the agreement between a candidate sampler and a reference method sampler.

The Comité Européen de Normalisation Standard EN 481 (CEN, 1993) describes size fraction definitions for workplace aerosol sampling, and identifies inhalable “conventions” relative to thoracic, respirable, extra-thoracic, and tracheobronchial penetration (but not necessarily deposition) in the respiratory system. They define a thoracic cumulative lognormal distribution with a MMD of 11.64 μm and a GSD of 1.5, such that 50% of airborne particles with a diameter of 10 μm are deposited in the thoracic region. The concept of using a pre-separator that has the same performance (penetration) characteristics as portions of the respiratory system have been discussed by a number of researchers, including Marple and Rubow (1976), Lippmann and Chan

(1979), Vincent and Mark (1981), Soderholm (1989), Liden and Kenny (1991), and John and Wall (1983). Watson et al. (1983), Wedding and Carney (1983), and Van der Meulen (1986) mathematically evaluated inlet design parameters in terms of collection efficiency relative to proposed sampling criteria. These reports suggest that factors such as extreme wind speed and coarse particle concentration could pose significant problems in meeting performance specifications.

A sampler's performance is generally described by a cumulative lognormal distribution. The cumulative lognormal distribution is defined by two characteristics: the cutpoint ($d_{50}$) and the slope. The cutpoint is the particle diameter that corresponds to the 50th percentile of the distribution. The slope is the ratio of the 84.1th percentile divided by the 50th percentile, the 50th percentile divided by the 15.9th percentile, or the square root of the 84.1th percentile divided by the 15.9th percentile. This cumulative lognormal distribution is referred to as the fractional efficiency curve of the sampler.

The ultimate goal of a PM sampler is to accurately measure the particle sizes that exist in the atmosphere. However, it is not currently possible to accurately characterize the material that exists as particles in the atmosphere because of difficulties in creating a reference standard for particles suspended in the atmosphere. No calibration standards for suspended particle mass exist; therefore, accuracy of particle mass measurements cannot be determined. As a result, the EPA defines accuracy for PM measurements in terms of the agreement between a candidate sampler and a reference sampler under standardized conditions for sample collection, storage, and analysis (USEPA, 1996a, 2001a). Therefore, sampler comparisons become very important in determining the reproducibility of sampler measurements (measurement precision, as defined by EPA) and how sampler designs influences accuracy (USEPA, 2001a). When using different measurement techniques, samplers of different design or manufacture and in some cases when using identical systems of different age or cleanliness, substantial biases of 50% or more have been observed. Regulatory and performance issues for the primary size selective PM samplers and related methods currently used in the scientific and regulatory realms (TSP, $PM_{10}$, and $PM_{2.5}$) are discussed in greater in the following sections.

### Total Suspended Particulate (TSP) Sampler

The TSP high-volume (HiVol) sampler has remained essentially unchanged since the sampler's identification as a reference ambient sampling device in 1971 (*Federal Register*, 1971). Wedding et al. (1977) reported that the TSP sampler's gable roof, used as a weather shield, removed a significant portion of the particles larger than 50 μm. McFarland and Ortiz (1979) reported that the cutpoint of the HiVol TSP sampler fluctuated with wind speed and direction and may vary from 25 to 40 μm AED. McFarland et al. (1979) reported that the slope of the TSP sampler ranged from 2.2 to 2.5, depending on the wind speed. Only minor technical updates have been incorporated in commercially available units, such as the types of available sequence and elapsed timers (mechanical, electronic) and the types of flow controllers (mass flow, volumetric) (USEPA, 1996a).

### Ambient $PM_{10}$

Not all countries categorize $PM_{10}$ samplers in the same manner. For instance, in the United States a $PM_{10}$ sampler is classified as having a penetration curve with a cutpoint of 10 μm while other countries (e.g. Japan) classify a $PM_{10}$ sampler as rejecting (removing from the air stream) all particles greater than 10 μm (USEPA, 2003). A significant step in the standardization process of aerosol sampling was the EPA definition (USEPA, 1987b) of the $PM_{10}$ size fraction, based on the AED of particles capable of penetrating to the thoracic region of the respiratory system. This definition was followed by the implementation of EPA's $PM_{10}$ Ambient Air Monitoring Reference and Equivalent Methods regulation. The Equivalent Method regulation format included the adoption of performance specification for aerosol samplers based on controlled wind tunnel testing with mono-dispersed aerosols (USEPA, 1996a). Ambient $PM_{10}$ samplers can be standalone units, as the shown in Figure 3, or attachment inlets as shown in Figure 4. The pre-separator for the ambient PM samplers employ impactors (Liu and

Pui, 1981; McFarland and Ortiz, 1982; Kim et al., 1998), as shown in Figure 4, or cyclones (Wedding et al., 1982) to limit particle collection.

$PM_{10}$ samplers are designated by EPA as reference or equivalent methods under the provisions of 40 CFR, Part 53 (*CFR*, 2001a). $PM_{10}$ reference methods must use the measurement principle and meet additional specifications set forth in 40 CFR, Part 50, Appendix J (*CFR*, 2001e). Reference method $PM_{10}$ samplers must also meet the requirements specified in 40 CFR, Part 53, Subpart D. Appendix J specifies a measurement principle based on extracting an air sample from the atmosphere with a sampler that incorporates inertial separation of the $PM_{10}$ size range particles followed by collection of the $PM_{10}$ particles on a filter over a 24-hour period. Alternatively, equivalent $PM_{10}$ methods are not required to conform to the measurement principle specified in Appendix J or meet the additional Appendix J requirements (USEPA, 1996a). Instead, equivalent $PM_{10}$ methods must meet the performance specifications set forth in 40 CFR, Part 53, Subpart D and demonstrate comparability to a reference method as required by 40 CFR, Part 53, Subpart C.

To determine the acceptability of the sampling effectiveness of the candidate sampler, the collection efficiency curve of the candidate sampler is compared to that of a specified “ideal” sampler. The model for this hypothetical “ideal” sampler, designed to mimic particle penetration to the thoracic region of the human respiratory tract is based on Chan and Lippman’s (1980) regression equation for extrathoracic deposition in the respiratory tract during mouth breathing. However, the “ideal” sampler’s penetration curve is sharper than the thoracic penetration curve (ACGIH, 1994; ISO, 1993; CEN, 1993). According to the USEPA (2001a, 2003), a $PM_{10}$ sampler with a penetration curve sharper than the thoracic curve has the advantage of reducing the problem of maintaining the finite collection efficiency specified by the thoracic curve for particles larger than 10 μm AED.

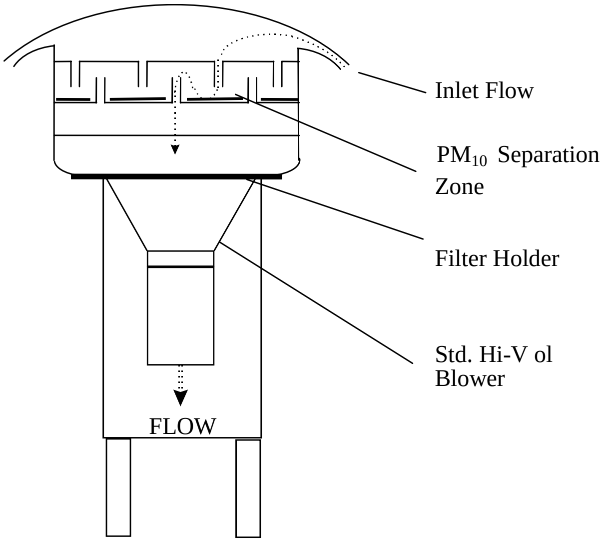


**Figure 3.** Graseby Andersen $PM_{10}$ sampler (Buch, 1999).

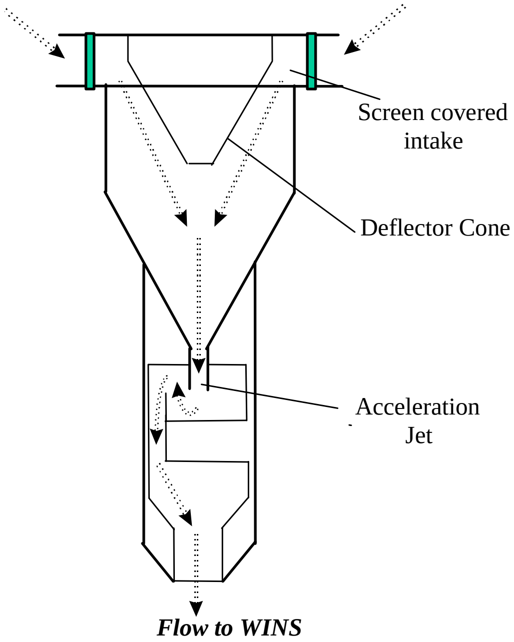


**Figure 4.** Graseby Model SA246A $PM_{10}$ “low flow” inlet (Buch, 1999).

40 CFR, Part 53, Subpart D describes the procedures for testing the performance characteristics of candidate $PM_{10}$ ambient air samplers.  In the full wind tunnel test, the candidate sampler's collection efficiency is determined for several mono-disperse particle sizes (i.e., liquid particle target diameters of 3, 5, 7, 9, 10, 11, 13, 15, and 20 μm AED) at wind speeds of 2, 8, and 24 km/h (*CFR*, 2001a).  A smoothed collection efficiency curve is generated using the individual collection efficiencies determined in the wind tunnel tests.  The candidate sampler's collection efficiency curve, along with the idealized ambient particle size distribution, is then used to determine the expected mass concentration for the candidate sampler.  The candidate sampler passes the liquid particle sampling effectiveness test if the expected mass concentration calculated for the candidate sampler, at each wind speed, differs by no more than +/- 10% from that predicted for the "ideal" sampler.  The candidate method passes the 50% cutpoint test if the resulting cutpoint at each wind speed falls within 10 +/- 0.5 μm.  The candidate sampler must also pass other tests listed in 40 CFR, Part 53, Subpart D; however, the full wind tunnel test is the primary test evaluating the sampler collection efficiency curve.  Additional information on conducting wind tunnel evaluations on $PM_{10}$ inlets was described by John and Wall (1983) and Ranade et al. (1990).

A number of samplers have been designated as $PM_{10}$ reference or equivalent method samplers (USEPA, 2001b).  Mass concentration measurements with a reproducibility close to 10% have been obtained with collocated samplers of identical design (USEPA, 1996a).  However, field studies of collocated EPA approved $PM_{10}$ samplers have shown substantial errors under certain conditions.  These errors result from: 1) allowing a tolerance of +/- 0.5 μm for the 10 μm cutpoint; 2) cutpoint deviations, beyond the established tolerances, associated with various field application parameters; 3) inadequate restrictions on internal particle bounce; 4) surface overloading; 5) soiling of certain types of $PM_{10}$ inlets; and 6) losses of semivolatile components.  According to the USEPA (1996a), the most significant performance flaws have combined to produce excessive (up to 60%) mass concentration errors.

Watson et al. (1983) affirmed that EPA's $PM_{10}$ performance specifications allowed a cutpoint tolerance range that could allow inlets to be "fine tuned", suggesting that the cutpoint could be adjusted to the lower or upper end of the range to suit particular sampling needs. For example, a "reduction" in reported concentration could be achieved by simply using a lower (e.g., 9.5 μm) cutpoint inlet that is still within the acceptable cutpoint range. The errors between acceptable samplers have been apparent in the data from sampler comparison studies (e.g., Rodes et al., 1985, Purdue et al., 1986; Thanukaos et al., 1992). Most of the reported errors between samplers were less than 10%, although some differences greater than 30% were reported. The reports suggest that the collection efficiency of high volume $PM_{10}$ sampler inlets based on cyclonic separation (Wedding, 1985) were consistently lower, while those based on low velocity impaction (McFarland et al., 1984) were consistently higher.

Wang and John (1988) were critical of the EPA's $PM_{10}$ performance specification on allowable particle bounce (*Federal Register*, 1987), stating that the criteria can lead to a 30% overestimation of mass under worst case conditions. In a related paper, John et al. (1991) reported that although reentrainment of particles deposited in a sampler inlet by airflow alone, is typically negligible, reentrainment caused from subsequent particle deagglomeration caused by "bombardment" can be substantial. John and Wang (1991) suggested that particle loading on oiled deposition surfaces can affect particle collection and strongly suggested that periodic cleaning and re-oiling should be required for $PM_{10}$ inlets. Vanderpool et al. (2001a) states that "particle bounce at an impaction surface occurs when the collection surface is unable to completely absorb the kinetic energy of the incident particle". Vanderpool et al. (2001a) further states that "if this inelastic collision occurs, the particle is not retained by the surface and can bias the size distribution measurement towards smaller aerodynamic sizes". In addition, overloading can occur when the layers of previously collected particles adversely change the nature of the collection surface (Vanderpool et al., 2001a).

Shifts in sampler cutpoints, attributed to soiling, have also been reported for cyclonic separators. Blachmann and Lippman (1974) reported that the performance of a

10 μm nylon cyclone was affected by loading, and the accumulation of particle deposits increased the collection efficiency (i.e., reduced the cutpoint). Tsai et al. (1999) determined that the penetration efficiency for a 10 μm cyclone was reduced from 97% to 71% for 3.06 μm diameter particles after a 0.4 mg loading.

Rodes et al. (1985) conducted a field comparison study and reported that the SA-321A $PM_{10}$ ambient air sampler collected an average of 0.3% less $PM_{10}$ and the WA-40CFM $PM_{10}$ ambient air sampler collected an average of 3.3 % more $PM_{10}$ than was present in the ambient air, as sampled by wide range aerosol classifier (WRAC). Rodes et al. (1985) stated that these estimates were more a measure of inlet performance "predictability" than measures of the error. Wedding et al. (1985) stated that the WRAC system, as used in the Rodes et al. (1985) field comparison study, was not satisfactory for obtaining particle size distributions. Rodes et al. (1985) also conducted wind tunnel studies and reported an average cutpoint of 6.6 μm AED for a dirty or used WA-40CFM sampler and an average cutpoint of 8.0 μm AED for a dirty or used SA-321A sampler.

Purdue et al. (1986) also compared the WA-40CFM and SA-321A samplers and reported variable concentration results between a new and used WA-40CFM sampler; similar results were reported for the SA-321A. The Andersen SA-321A $PM_{10}$ sampler was found to collect an average of 58% more mass than a collocated Wedding $PM_{10}$ sampler. This was partly attributed to the predicted error associated with cutpoint differences between the inlets. A more significant error (not predicted) was associated with degraded performances in opposite directions (Andersen over-sampling, Wedding under-sampling) because of soiling of the separators during extended sampling periods. Purdue et al. (1986) also observed variable results between the SA-321A and WA-40CFM samplers when both were tested at the same location. Purdue et al. (1986) did not measure the PSD of the dust being sampled, giving no indication of the samplers performance characteristics.

Sweitzer (1985) reported that there was a 15% variation between the SA-321A and WA-40CFM samplers, with the SA-321A sampler providing consistently higher values. Herber (1998) conducted a property line sampling study at two stripper cotton gins in Texas using TSP samplers and two $PM_{10}$ style samplers (WA-40CFM and SA1200 $PM_{10}$ inlets). Herber (1998) reported the WA-40CFM sampler measured 62.4% of the actual $PM_{10}$ mass concentration and the SA-1200 sampler measured 1.1 times the actual $PM_{10}$ mass concentration.

Ranade et al. (1990) evaluated two high-volume $PM_{10}$ sampler inlets, the Sierra Andersen Model 321A (SA-321A) and the Wedding IP10, using EPA's sampler performance testing methods. Ranade et al. (1990) reported that SA-321A had a cutpoint of 10.5 μm and a slope of 1.4 (liquid particles) and a cutpoint of 11.1 μm and a slope of 1.46 (solid particles) at a wind speed of 8 km/h. The Wedding IP10 was reported to have a cutpoint of 9.5 μm and a slope of 1.32 (liquid particles) and a cutpoint of 9.6 μm and a slope of 1.35 (solid particles) at a wind speed of 8 km/h. Tests conducted at a wind speed at 2 km/h showed that the SA-321A sampler had a cutpoint of 10.7 μm and a slope of 1.42 (liquid particles) and a cutpoint of 10.6 μm and a slope of 1.49 (solid particles). The Wedding IP10 had a cutpoint of 9.6 μm and a slope of 1.27 (liquid particles) and a cutpoint of 9.65 μm and a slope of 1.33 (solid particles) at a wind speed of 2 km/h.

Ono et al. (2000) reported on a study using a Partisol, TEOM, dichotomous, Wedding high-volume sampler, and the Graseby high-volume $PM_{10}$ samplers, which were collocated and operated at a location with high concentrations of coarse PM. Ono et al. (2000) reported that TEOM and Partisol samplers agreed to within 6% on average; however, the dichotomous, Graseby, and Wedding samplers measured significantly lower $PM_{10}$ concentrations than the TEOM (on average, 10, 25, and 35% lower, respectively). Ono et al. (2000) attributed these lower concentrations to a decrease in cutpoint caused by wind speeds and cleanliness of the inlet.

Wang et al. (2003) evaluated Graseby-Andersen FRM $PM_{10}$ samplers in a dust chamber where the samplers were exposed to treatments of dispersed cornstarch, fly ash, and aluminum oxide. Wang et al. (2003) reported that the Graseby-Andersen FRM $PM_{10}$ sampler over-sampled the dispersed cornstarch, fly ash, and aluminum oxide by an average of 89%, 41%, and 14%, respectively. Wang et al. (2003) also reported that the average cutpoint and slope for the Graseby-Andersen sampler was 12.5 μm and 1.3 when sampling cornstarch; 17.7 μm and 1.5 when sampling fly ash; and 17 μm and 1.5 when sampling aluminum oxide. Wang et al. (2003) concluded that the Graseby-Andersen FRM $PM_{10}$ sampler's fractional efficiency curve shifted to the right when sampling dust with smaller MMDs.

### Ambient $PM_{2.5}$

The FRM $PM_{2.5}$ samplers aspirate air from the atmosphere at 16.7 lpm through an inlet specifically designed to be insensitive to wind speed and direction and reject insects and precipitation. A schematic of a FRM sampler is shown in Figure 5. The FRM sampler consists of two pre-separators. The initial (inlet) pre-separator is an impactor designed to remove particles larger than a nominal 10 μm AED from the sampled air. A schematic of the $PM_{10}$ inlet is shown in Figure 4 (Tolocka et al., 2001). The second pre-separator (originally the Well Impactor Ninety-Six (WINS)) is located downstream of the inlet and is designed to remove particles greater than a nominal 2.5 μm, allowing the remaining PM to be collected on a Teflon filter (Peters et al., 2001b). A schematic of the WINS impactor is shown in Figure 6. A cyclonic separator, the Sharp Cut Cyclone (SCC), was designed as a substitute for the WINS impactor for $PM_{2.5}$ sampling. A schematic of the SCC is shown in Figure 7.

The Federal Reference Method (FRM) $PM_{2.5}$ samplers are specified by design, unlike the performance based FRM standard for the $PM_{10}$ samplers. An update published by the USEPA (2000) states: "the requirement that these instruments rely on specific design elements, rather than performance criteria alone, is structured to produce greater measurement reproducibility and to avoid the data measurement uncertainties experienced in the $PM_{10}$ monitoring program."

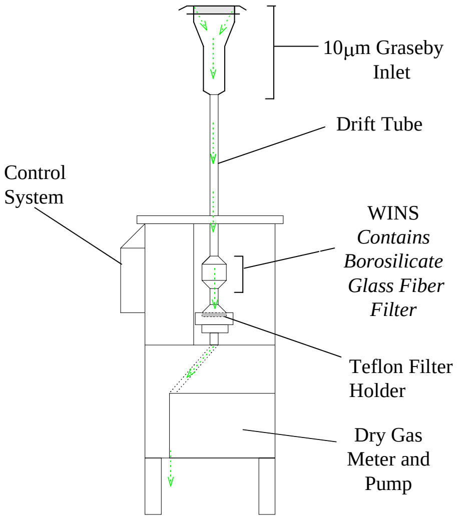


**Figure 5.** Graseby Andersen FRM $PM_{2.5}$ sampler (Buch, 1999).

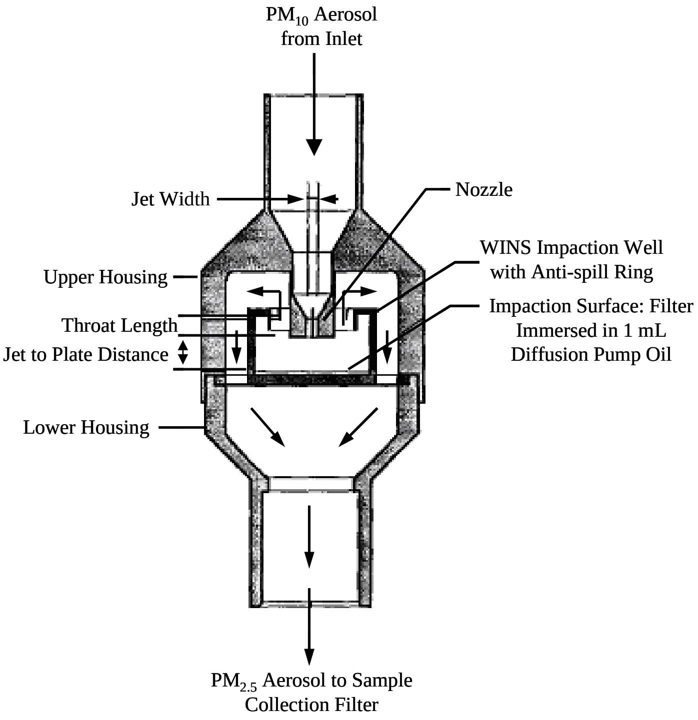


**Figure 6.** WINS separator (Vanderpool et al., 2001b).

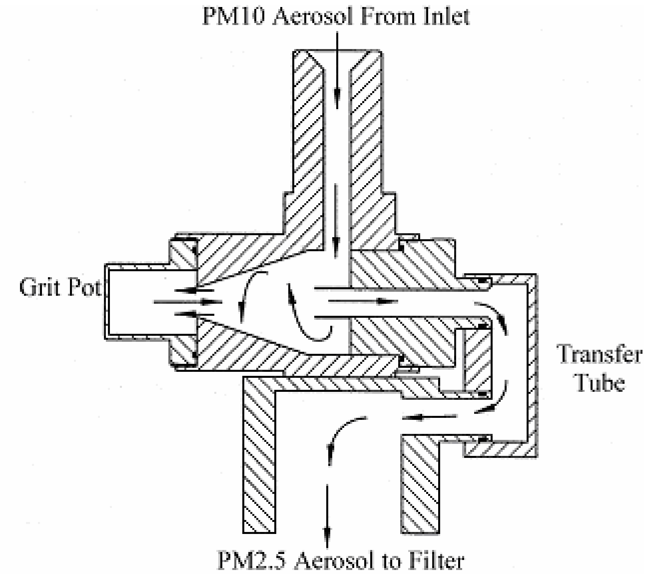


**Figure 7.** Sharp cut cyclone (Pargmann, 2001).

In addition to FRM $PM_{2.5}$ sampler designation, the EPA also provides a Federal Equivalent Method (FEM) $PM_{2.5}$ sampler designation. The EPA defined three FEM classes (Class I, Class II, and Class III) based on the degree of dissimilarity between a candidate sampler and the FRM requirements (*CFR*, 2001d). An increase in equivalency designation, from Class I to Class II to Class III, indicates a greater deviation from the FRM, requiring more extensive testing for equivalency verification. Class I equivalent methods correspond to candidate samplers that have only minor deviations from the reference method, usually relating to sample transmission component modifications incorporated to accommodate a sequential sampling mechanism. A Class I FEM candidate sampler must undergo the same testing as the FRM candidate sampler, with the addition of an internal aerosol transport test. Class II equivalent methods are 24-hour integrated filter collection techniques that rely on gravimetric analysis, but have significant design or performance deviations from the reference method. For example, substituting a cyclone separator for the WINS is a deviation from the FRM that could be designated as Class II FEM. A Class II FEM candidate sampler must undergo more extensive testing than the FRM or Class I FEM, with the tests being specific to the nature of the modifications in the candidate method. Additional testing may include all, or some subset, of the following tests: full wind tunnel test, wind tunnel aspiration test, static fractionator test, loading test, and volatility test. Class III equivalent methods do not fall under Class I or Class II designation because of further deviations from the FRM, but still provide mass concentration measurements of $PM_{2.5}$ comparable to the reference method. The two primary sampling categories that fall into this class are non-filter-based techniques and continuous (or semi-continuous) analyzers. Specific requirements for Class III FEM are not defined because of the wide range of technologies that might be employed for $PM_{2.5}$ mass measurement. As a result, specific Class III FEM testing and other requirements are developed by EPA on a case-by-case basis. Class III FEMs may be required to undergo any or all of the testing required for validation as an FRM, Class I FEM, or Class II FEM, as well as additional testing specific to the sampling technology.

FRM samplers are defined by the design. The basic design of the FRM sampler is given in the Federal Register (1997) and 40 CFR, Part 50, Appendix L (*CFR*, 2001d). Performance specifications for FRM samplers are listed in 40 CFR, Parts 53 and 58 (*CFR*, 2001 a, b). The accuracy of FRM sampler is determined through collocated sampler evaluation tests. The performance specification for FEM Class I samplers are very similar to those required for FRM sampler. Detailed performance specifications are listed in 40 CFR, Part 53. A candidate $PM_{2.5}$ sampler classified as a Class II FEM is required to meet a more rigorous set of performance criteria, as defined in 40 CFR, Part 53.

40 CFR, Part 53, Subpart F describes the procedures for testing the performance characteristics of Class II FEM candidate $PM_{2.5}$ ambient air samplers. In the full wind tunnel test, the candidate sampler's collection efficiency is determined for several mono-disperse particle sizes (i.e., solid particle target diameters of 1.5, 2.0, 2.2, 2.5, 2.8, 3.5, and 4.0 μm AED) at wind speeds of 2 and 24 km/h (*CFR*, 2001a). A smooth collection efficiency curve is then generated using the individual collection efficiencies determined

in the wind tunnel tests. The candidate sampler's collection efficiency curve, along with the three idealized ambient particle size distributions (coarse, "typical" coarse, and fine), is then used to determine the expected mass concentration for the candidate sampler. The candidate sampler passes the full wind tunnel evaluation if the expected mass concentration calculated for the candidate sampler, at each wind speed and for each idealized distribution, differs by no more than +/- 5% from that predicted for the "ideal" sampler. The candidate method passes the 50% cutpoint test if the test result at each wind speed falls within 2.5 +/- 0.2 μm. The candidate sampler must also pass the wind tunnel aspiration, static fractionator, loading, and volatility tests listed in 40 CFR, Part 53, Subpart F; however, the full wind tunnel test is the primary test evaluating the samplers collection efficiency curve.

Vanderpool et al. (2001b) listed several factors that influence the mass concentration measured by the FRM WINS sampler including: PM concentration and size distribution; chemical composition of the collected aerosol; sampler volumetric flow rate (affected by the accuracy of the sampler's ambient temperature, ambient pressure, and flow sensors); sampling time; sampler inlet geometry; performance of the sampler's internal size-selective separator; sampler internal particle losses; pre-sampling and post-sampling filter conditioning; and all other associated sampling and analysis procedures. In addition, relatively small changes in a sampler's cutpoint can produce a significant and hard to predict mass concentration errors (USEPA, 1996a). Therefore, factors that affect sampler concentration errors should be identified and the corresponding influences determined as a function of particles size.

According to Vanderpool et al. (2001b), "Regardless of the inertial fractionation mechanism (conventional impaction, virtual impaction, or cyclonic separation) and the separator design, all separators overload to some degree if continuously exposed to particle-laden airstreams". One method of determining the sampler uncertainty attributed to overloading is to evaluate the elemental composition of $PM_{2.5}$ and $PM_{10}$, or the coarse fraction of $PM_{10}$. Using this method, elements relating to soil type materials have been found in the $PM_{2.5}$ fraction. In a study using dichotomous samplers, the soil type material found in the $PM_{2.5}$ fraction was equivalent to 5% of the coarse mode fraction of $PM_{10}$ (Dzubay et al., 1988). Similar results were reported from the IMPROVE network, which suggested that the soil derived material found in the $PM_{2.5}$ sample was equivalent to 20% of the coarse fraction of $PM_{10}$ (Eldred et al., 1994).

Pitchford (1997) stated that an early concern with WINS impactor was cleaning to avoid the possibility of having part of the impactor deposit break off and make its way to the filter, thereby giving a falsely high measurement of $PM_{2.5}$. Pitchford (1997) reported that a dirty WINS impactor tended to produce a falsely low measurement of $PM_{2.5}$. Pictchford (1997) suggested that this falsely low measurement could be attributed to deposits building up on the impaction surface, in effect changing the critical dimensions of the WINS, resulting in a low cutpoint. Vanderpool et al. (2001a) evaluated the loading characteristics of the WINS separator by monitoring the sampler's performance after repeated operation in an artificially generated, high concentration, coarse mode aerosol composed of Arizona Test Dust, as well as in field tests. In the wind tunnel experiments, the WINS performance was found to be a monotonic function of loading. A negative 5% error in the $PM_{2.5}$ measurement resulted from a coarse particulate loading of approximately 16 mg because of a slight reduction in the separator's cutpoint. It was also determined that the results from the laboratory experiments could not be extrapolated to the field settings and that the performance of the WINS was more sensitive to impactor loading in the field tests than in experiments with the single component aerosol.

Kenny et al. (2000) evaluated a clean WINS, SCC, GK, and University Research Cyclone (URG) using EPA's procedures for testing the performance characteristics of Class II equivalent $PM_{2.5}$ methods. Kenny et al. (2000) reported that the SCC could over-sample “coarse” aerosols by 4 to 5%. The URG cyclone could over-estimate “coarse” aerosols by more than 13% and the GK could over-estimate “coarse” aerosols by more than 9%. Kenny et al. (2000) also reported that the clean WINS impactor was within 1% of the ideal concentration, which was expected since the ideal penetration curve is a sigmoid model fit to the WINS impactor data.

The WINS impactor was designed to be deployed downstream of the Graseby-Anderson 246A $PM_{10}$ inlet and operate at a flow rate of 16.7 lpm. Peters and Vanderpool (1996), under contract with EPA to evaluate the WINS sampler, characterized the WINS penetration curve as having a cutpoint of 2.48 μm AED and a slope of 1.18. Peters et al. (2001c) evaluated the WINS using mono-disperse aerosols and reported that the WINS cutpoint ranged from 2.44 to 2.48 μm and the slope of the sampler's penetration curve ranged from 1.17 to 1.22. Vanderpool et al. (2001b) stated that "unlike conventional greased flat plate impactors, the general effect of loading in the WINS separator is to reduce the cutpoint rater than to increase it". Vanderpool et al. (2001b) reported that the cutpoint for 13 archived WINS samplers from the various field sites after 5 days of loading ranged from 2.32 μm to 2.51 μm.

Kenny (1998) conducted an evaluation study on the WINS impactor, SCC, GK4.39 cyclone, and the URG. The SCC was based on the design of the SRI Cyclone III described by Smith et al. (1979) and the URG cyclone was based on the Stairmand design evaluated by Moore and McFarland (1993). Kenny (1998) reported cutpoints (slopes) of 2.44 μm (1.23), 2.46 μm (1.19), 2.37 μm (1.28), and 2.46 μm (1.45), respectively, for the WINS, SCC, GK4.39, and the URG samplers using mono-disperse particles. Kenny et al. (2000) evaluated the WINS and SCC when loaded with Aloxite dust (and no $PM_{10}$ inlet) and determined that the WINS cutpoint shifted steadily downwards to 2.15 μm, whereas the SCC cutpoint did not exhibit a significant downward shift.

Buch (1999) evaluated the WINS and the IMPROVE $PM_{2.5}$ samplers in a dust chamber using poly-disperse particles. Buch (1999) determine that the WINS cutpoint was 2.7 +/- 0.41 μm and the slope was 1.32 +/- 0.03 when exposed to a dust consisting of 67% $PM_{2.5}$. The IMPROVE $PM_{2.5}$ sampler was reported to have an average cutpoint of 3.8 μm and an average slope of 1.23 (Buch, 1999). Pargmann (2001) conducted a similar study that evaluated WINS, SCC, and the hi-vol $PM_{2.5}$ sampler in a dust chamber using poly-disperse particles (i.e., Alumina, corn starch, and wheat flour). No cutpoints or slopes were reported for the SCC or hi-vol $PM_{2.5}$ samplers; however, the WINS fractional efficiency curve was defined by a cutpoint of 1.95 +/- 0.10 μm and a slope of 1.31 +/- 0.04 when exposed to a dust consisting of 5.34% $PM_{2.5}$. Pargmann (2001) also reported the percent error between the sampler measurements and actual $PM_{2.5}$ concentrations. The WINS sampler over-sampled by 51%, 211%, and 444% when sampling Alumina, corn starch, and wheat flour, respectively. The SCC sampler over-sampled by 119%, 585%, and 1,771%, when sampling Alumina, corn starch, and wheat flour, respectively. The hi-vol $PM_{2.5}$ sampler over-sampled by 111%, 467%, and 632% when sampling Alumina, corn starch, and wheat flour, respectively. Pargmann (2001) stated that sampler over-sampling increased as the MMD of dust being sampled increased.

BGI Incorporated developed the Very Sharp-Cut Cyclone (VSCC) that was based on the design of the SCC described by Kenny et al. (1998). The VSCC differs from the SCC in that it has a longer cone, wider base diameter, and decreased inlet and outlet tube diameters. The evaluation study conducted by Kenny (2000) consisted of testing the VSCC and the WINS impactor in a wind tunnel using solid, spherical glass microspheres (density of 2.45 $g/cm^3$) with physical diameters up to 25 μm (MMD of the test aerosol was 4 μm) at a loading rate of 100 to 200 $particles/cm^3$. Kenny (2000) reported cutpoints (slopes) of 2.48 μm (1.22) and 2.5 μm (1.157) for the WINS impactor and VSCC (operated at 16.67 lpm), respectively.

Peters et al. (2001a) evaluated the SCC 1.829, SCC 2.141, and AN 3.68 $PM_{2.5}$ cyclones and a Spiral impactor using the EPA procedures for testing the performance characteristics of Class II equivalent $PM_{2.5}$ methods. Each of these cyclones separators are based on the SRI designs described by Smith et al. (1979). Peters et al. (2001a) reported a cutpoint of 2.44 μm and a slope of 1.23 for the SCC 1.829. The SCC 2.141 was reported to have a cutpoint of 2.52 μm and 2.35 μm for flow rates of 6.7 and 7.0 lpm, respectively. The slope associated with the SCC 2.141 was reported as 1.24 for both flow rates tested. Peters et al. (2001a) reported that the SCC 2.141 over-estimated the idealized “coarse” mass concentration by as much as 6.1% at a flow rate of 6.7 lpm. The AN3.68 was reported to have a cutpoint of 2.72 μm and a slope of 1.15 when operated at the design flow rate of 24.0 lpm. Peters et al. (2001a) reported that the AN 3.68 over-estimated the idealized "coarse" mass concentration by 7.4%, which was attributed to the sampler's larger cutpoint. Peters et al. (2001a) reported that cutpoint associated with the Spiral impactor was highly variable and ranged from 1.9 to 2.7 μm for three separate tests when operated at the design flow rate of 7.0 lpm. Peters et al. (2001a) characterized the performance of the ungreased Spiral impactor by a cutpoint of 2.69 μm and a slope of 1.30. Kenny et al. (2000) concluded that cyclonic separators become more efficient with increased loading (i.e., the cutpoint shifts to the left with increased loading).

The MiniVol, designed to have a 2.5 μm AED cutpoint at a flow rate of 5 lpm, does not meet the design specifications required for designation as a $PM_{2.5}$ regulatory monitor (Hill et al., 1999). Based on the data provided by Hill et al. (1999) the MiniVol 2.5 μm impactor appeared to have a cutpoint of 2.7 μm and a slope of 1.4 when wind tunnel tested using mono-disperse particles. Hill et al. (1999) also evaluated a MiniVol $PM_{2.5}$ impactor with various impactor plate grease loadings. The MiniVol impactor appeared to have a cutpoint ranging from 2.66 to 2.82 μm with a slope ranging from 1.25 to 1.37 based on data provided by Hill et al. (1999) for a wind tunnel study using mono-disperse particles and various application rates (defined as light, heavy, and very heavy) of grease on the impactor plate. Hill et al. (1999) also noted that recent modifications of the MiniVol $PM_{2.5}$ impactor design required the use of a $PM_{10}$ impactor upstream of the $PM_{2.5}$ impactor (i.e., cascade or tandem impactor configuration). Hill et al. (1999) provided data that was used to estimate the cutpoint (and slopes) associated with the MiniVol $PM_{2.5}$ impactor using a flat plate, cup plate, flat plate following a $PM_{10}$ impactor, and a cup plate following a $PM_{10}$ impactor that were determined to be 2.7 μm (1.48), 2.97 μm (1.29), 2.7 μm (1.65), and 3.1 μm (1.29), respectively.

The EPA staff recommended the use of a sharp 2.5 μm cutpoint for a fine particle indicator (USEPA, 1996a). However, $PM_{2.5}$ samplers do have some potential for an intrusion of the "tail" of the coarse mode during episodes of fugitive dust concentrations. The EPA staff recommends a sharp inlet for the FRM to minimize this potential intrusion of coarse mode particles. According to USEPA (1996a), "Such intrusions into $PM_{2.5}$ measurement are not anticipated to be significant in most situations. Nevertheless, if subsequent data reveal problems in this regard, this issue can, and should be, addressed on a case-by-case basis in the monitoring and implementation programs. Because the purpose of a $PM_{2.5}$ standard is to direct controls toward sources of fine mode particles, it would be appropriate to develop analytical procedures for identifying those cases where a $PM_{2.5}$ standard violation would not have occurred in the absence of coarse mode particle intrusion. Consideration should be given to a policy similar to the natural events policy for addressing such cases."

The available data show that typically only 5-15% (on the order of 1 to 5 $\mu g/m^3$) of the $PM_{2.5}$ mass is attributable to soil-type sources even in dusty areas such as San Joaquin Valley, California, and Phoenix, Arizona (USEPA, 1996a). However, this percentage may increase during events such as high winds. According to USEPA (1996a), "A sharper inlet for the Federal Reference Method may help to minimize the intrusion of coarse mode particles into the $PM_{2.5}$ measurement".

### Ambient $PM_{coarse}$

Currently, no consensus exists on the best technique for collecting $PM_{coarse}$. Potential methods for determining $PM_{coarse}$ include: multistage impaction, virtual impaction, and the difference method (i.e. subtracting $PM_{2.5}$ mass from $PM_{10}$ mass as determined from collocated $PM_{10}$ and $PM_{2.5}$ samplers) (USEPA, 2003). One problem associated with the difference method is that if either the $PM_{2.5}$ or $PM_{10}$ sampler fails, no $PM_{coarse}$ measurements can be calculated. In addition, errors associated with sampler cutpoints, flow rates, and filter weights (both before use and after collection and equilibration of particles) and errors attributed to loss of semivolatile components of PM may occur for each cut size. In general, most $PM_{coarse}$ data currently available and used by EPA and other institutions is based on the difference method.

The median $PM_{coarse}$ concentration across the United States during 1999, 2000, and 2001 was 10 $\mu g/m^3$, with a 95th percentile value of 21 $\mu g/m^3$ (USEPA, 2003). These estimates were based on the difference method and are subject to the effects of errors in measuring both $PM_{10}$ and $PM_{2.5}$. As a result of using the difference method, estimates of $PM_{coarse}$ concentrations have, at times, resulted in negative values based on currently available data (e.g., EPA AIRS Database). In addition, $PM_{coarse}$ was reported to be less uniform than $PM_{2.5}$ in most cities and crustal material was reported to be the primary constituent of $PM_{coarse}$, which are generally unlikely to exert notable health effects under most ambient exposure conditions.

Loo et al. (1979) reported that the cutpoint of the virtual impactor (dichotomous sampler) was 2.5 μm with a slope of 1.40. Multistage inertial impactors or cascade impactors provide discrete samples associated with selected particle size ranges that can be analyzed for mass or other constituents. Typically, these samplers have had uncharacterized inlets when used for ambient monitoring, which often results in a misinterpretation of the MMD with respect to total mass collected (USEPA, 1982a). Dzubay et al. (1979) reported that allowing entry of particles much larger than the first stage cutpoint could cause particles to bounce to lower stages, shifting the calculated MMD.

Marple et al. (1987) was first to report the calibration of the MS&T$^{TM}$ impactors and reported that the MS&T$^{TM}$ impactors produced $PM_{10}$ and $PM_{2.5}$ results that were comparable to the Dichotomous Sampler when operated at 4 lpm. Olson (1997) conducted a series of studies to determine the performance characteristics of the MS&T$^{TM}$ impactors when operated 10 and 20 lpm and exposed to mono-disperse particles. Olson (1997) reported that the performance characteristics of the MS&T$^{TM}$ $PM_{10}$ impactor could be described by a cutpoint of 10.3 μm and a slope of 1.10 when operated at 10 lpm (exposed to seven mono-disperse particle sizes ranging from 7.78 to 12.5 μm) and a cutpoint of 10.1 μm and a slope of 1.07 when operated at 20 lpm (exposed to seven mono-disperse particle sizes ranging from 8.3 to 11.0 μm). The performance characteristics of the MS&T$^{TM}$ $PM_{2.5}$ impactor, reported by Olson (1997), were described by a cutpoint of 2.52 μm and a slope of 1.07 when operated at 10 lpm (exposed to eight mono-disperse particle sizes ranging from 1.83 to 3.02 μm) and a cutpoint of 2.51 μm and a slope of 1.26 when operated at 20 lpm (exposed to seven mono-disperse particle sizes ranging from 1.81 to 3.46 μm). The slopes for each of the impactors were calculated as the square root of $d_{84.1}$ divided by $d_{15.9}$ using the data provided by Olson (1997).

Froines and Sioutas (2002) reported on the development and evaluation of a $PM_{10}$ Impactor-Inlet for a Continuous Coarse Particle Monitor used to acquire the coarse fraction of PM mass. The $PM_{10}$ inlet was operated at 50 lpm and was reported to have a cutpoint of 9.3 μm and a slope of 1.06. These performance characteristics were determined by subjecting the inlet to five PM size ranges (< 0.1 μm, 0.1 to 0.32 μm, 0.32 to 1.0 μm, 1.0 to 2.5 μm, and 2.5 to 10 μm) produced from three collocated Micro-Orifice Uniform Deposit Impactors (MOUDI). The inlet was evaluated at wind speeds of 3, 8, and 24 km/h in a wind tunnel and was reported to be statistically unaffected by the various wind speeds. The Continuous Coarse Particle Monitor was equipped with a virtual impactor and designed to have a theoretical cutpoint of 2.5 μm when operated at an intake flow rate of 50 lpm. In field studies, Froines and Sioutas (2002) reported that the Continuous Coarse Particle Monitor and the co-located Dichotomous Partisol-Plus (Model 2025 Sequential Air Sampler, Rupprecht and Patashnick Co. Inc., Albany, NY) produced very comparable results.

### Continuous PM Samplers

Long et al. (2002) discussed the need for continuous PM mass sampling techniques that could provide real time information on pollution levels and reduce the costs associated with traditional sampling techniques (e.g. costs associated with changing out filters and conditioning filters). Two methods of obtaining continuous PM mass sampling include the automated Tapered Element Oscillating Microbalance (TEOM) technology (Patashnick and Rupprecht, 1991) and the automated beta attenuation monitors (Merrifield, 1989; Wedding and Weigand, 1993).

The TEOM sampler computes mass based on the frequency shift as particles are deposited on an oscillating element. Patashnick and Rupprecht (1991) reported consistent and linear relationships between the TEOM and traditional gravimetric $PM_{10}$

samplers. Other studies (Cahill et al., 1994; Meyer et al., 1992; Meyer, et al., 1995) have reported that the modification of the aerosol by elevated operating temperatures appear to significantly effect the measured mass concentration. Cahill et al. (1994) reported that the TEOM sampler showed poor correlations and errors on the order of 30% lower than $PM_{10}$ gravimetric samplers in dry, dusty conditions. A WESTAR (1995) report concluded that on average the TEOM sampler concentrations were 21.8% lower than other collocated $PM_{10}$ samplers for concentrations greater than 50 $\mu g/m^3$.

The beta gauge mass monitor requires more frequent filter changes than the TEOM and is less sensitive to changes in mass caused by changes in relative humidity (USEPA, 2001a). Most beta gauge monitors heat the inlet, causing the evaporation of a substantial fraction of the particle bound water and an unknown fraction of the semivolatile PM. Arnold et al. (1992) reported that the Wedding beta gauge mass concentrations were 19% (on average) lower than collocated Wedding $PM_{10}$ gravimetric samplers. USEPA (1996a) stated that field tests indicated errors in the results of both the beta gauge and TEOM samplers when compared to gravimetric based samplers, which were not identified by the EPA performance test requirements.

### PM Stack Samplers

Emissions from stationary sources are determined primarily by stack sampling. A variety of techniques are available for the various pollutants of interest. All these techniques rely on measurements of stack flow rates and pollutant concentrations in order to determine the pollutant emissions rates. The original EPA method for determining PM emission rates was Method 5 (*Federal Register*, 1977), used to determine TSP emission rates through isokinetic stack sampling (USEPA, 1996a). In response to the 1987 NAAQS changes, EPA approved Method 201a. Method 201a is a constant sampling rate procedure (isokinetic) that utilizes a stainless steel cyclone to determine $PM_{10}$ emission rates from exhaust stacks. EPA is currently developing a new method (currently titled Pre-004) using a new cyclone with a nominal cutpoint of 2.5 μm in series with the Method 201a cyclone. This sampling system consists of a nozzle (matched with the air velocity in the stack to provide isokinetic sampling), $PM_{10}$ cyclone with a grit pot, $PM_{2.5}$ cyclone with a grit pot, and a filter holder that attaches to the Method 5 sampling train.  A picture of the combination $PM_{10}$ and $PM_{2.5}$ stack sampler is shown in Figure 8.

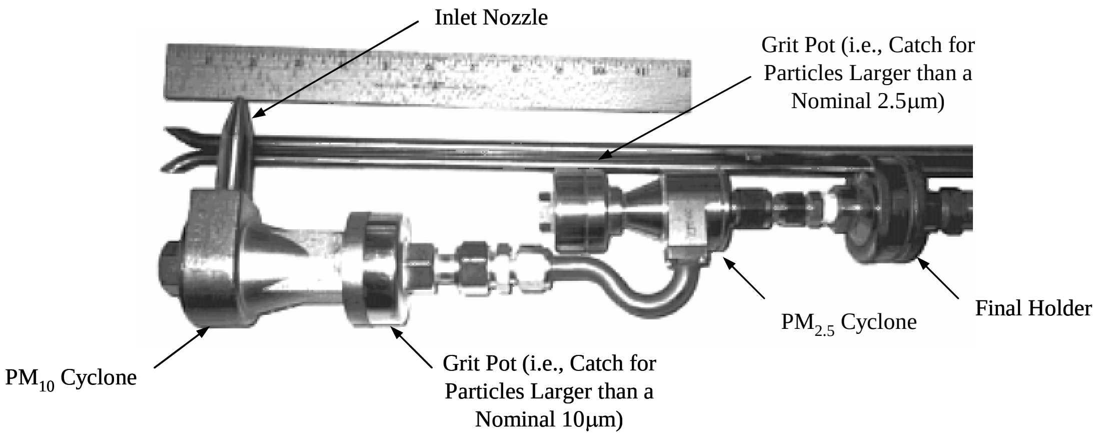


**Figure 8.** $PM_{10}$ and $PM_{2.5}$ cyclone combination sampler.

USEPA (2002) describes the validation methods and procedures and the criteria of acceptance for in-stack $PM_{10}$ samplers.  The operating principle of this in-stack sampler requires that isokinetic sampling be maintained within the well-defined limits, as deviations in the sampling flow rate can distort the flow pattern in the stack resulting in $PM_{10}$ measurement errors.  The validation methods call for the in-stack sampler to be tested in a wind tunnel at target gas velocities of 7 +/- 1.0, 15 +/- 1.5, and 25 +/- 2.5 m/s. The samplers collection efficiency is evaluated by exposing the sampler to dispersed concentrations of mono-disperse particles.  The various mono-disperse particle size used in the wind tunnel validation studies include: 5, 7, 10, 14, and 20 μm.  A smooth curve is drawn through the reported collection efficiencies, associated with the various mono-disperse particle sizes, and compared to the curves shown in Figure 9.  According to the USEPA (2002), the in-stack sampler's performance is acceptable if the reported fraction efficiency curve falls within the banded region for all particle sizes tested (shown in Figure 9) and the sampler's cutpoint is 10.0 +/- 1.0 μm AED.

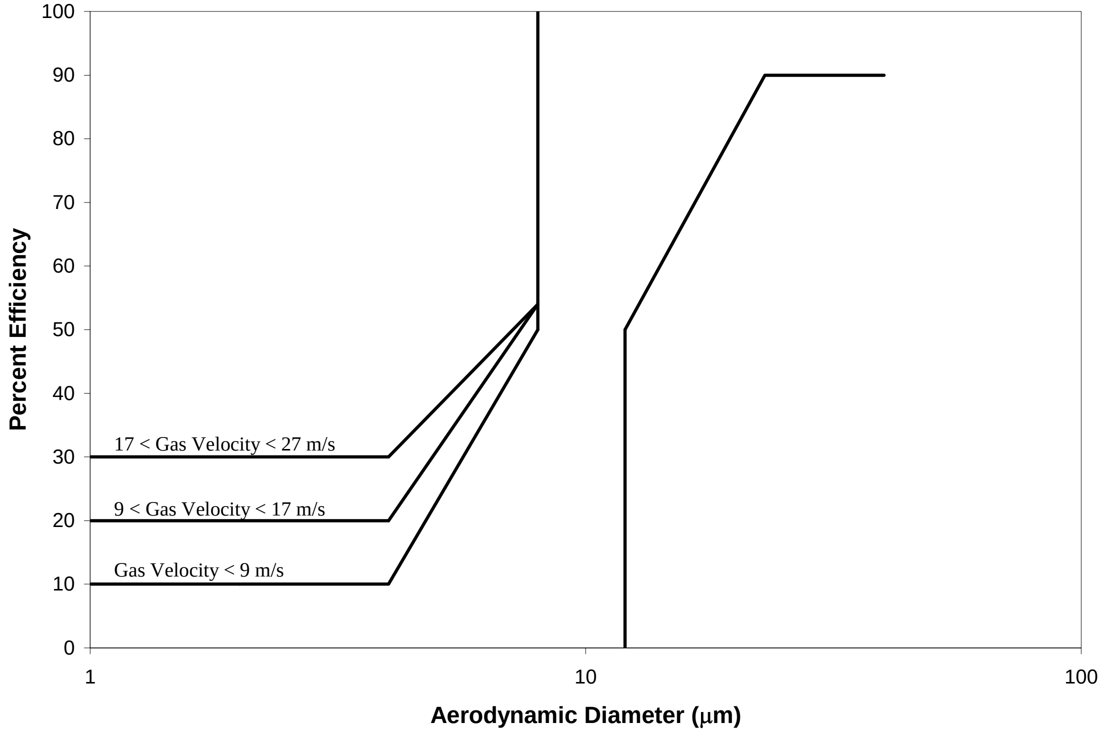


**Figure 9.** Efficiency envelope for the $PM_{10}$ cyclone (USEPA, 2002).

Literature pertaining to the performance requirements for EPA's $PM_{2.5}$ stack sampler used in Method Pre-004 is extremely sparse. The performance criteria for the $PM_{2.5}$ cyclone are essentially limited to a defined cutpoint diameter range. No slope or overall efficiency criteria are defined by USEPA (1999a). The required cutpoint diameter for the Method Pre-004 $PM_{2.5}$ cyclone is defined as 2.5 +/- 0.25 μm AED.

Smith et al. (1979) reported on the development and evaluation of five stage cyclone stack sampler design to operate at a flow rate of 28.3 lpm. In subsequent literature, the cyclones are referred to a Southern Research Institute (SRI) Cyclones I through V. The barrel diameters associated with the cyclones were 4.47, 3.66, 3.11, 2.54, and 1.52 cm, for Cyclones I through V respectively. Smith et al. (1979) calibrated the cyclones using mono-disperse aerosols over ranges in temperature, flow rate, and particle density and compared the results with a Climet Model 208A particle counter. Smith et al. (1979) reported cutpoints of 3.8 μm, 1.5 μm, 0.95 μm, 0.64 μm, and 0.32 μm for Cyclones I through V, respectively. Smith et al. (1979) provided collection efficiency curves for each of the cyclones, but no explicit slope values were reported. Based on the data provided by Smith et al. (1979), the cutpoints appeared to be sensitive to air temperature and flow rate (i.e. the cutpoint increased with increased temperature and the cutpoint increased with a decrease in flow rate).

According to John and Reischl (1980), the SRI stack-sampling cyclone was based on the T-2A cyclone designed by Chang (1974). John and Reischl (1980) developed a cyclone similar to the SRI cyclone, all critical cyclone dimensions being the same, that was designed to operate at a flow rate of 15 lpm. John and Reischl (1980) reported that flow rate greatly affected the cyclone's cutpoint (increased cutpoint with a decrease in flow rate) and slope (increase in slope with a increase in flow rate); the cutpoint was described by $52.5*Q^{-0.99}$. Other researchers (Bernstein et al., 1976; and Leith and Mehta, 1973) have used the same function with exponents ranging from 0.5 to 1.5 to describe a cyclone sampler's cutpoint with respect to flow rate. Bernstein, et al. (1976) reported a break in the cutpoint vs. flow rate curve for the Dorr-Oliver cyclone at a flow rate of 5 lpm. Earlier experiments by Blachman and Lippmann (1974) also reported breaks in the curve at 5 lpm where the Reynolds number at the inlet becomes greater than 2000.

Dahlin and Landham (2002) evaluated three of the Southern Research Institute cascade cyclones described by Smith et al. (1979). Dahlin and Landham (2002) reported that the MMD of the dust captured on the cyclone filters ranged from 10 to 15 μm, 6 to 8 μm, 4 to 5 μm, and 2.5 to 3.5 μm, respectively for no cyclone, Cyclone I alone, Cyclone II alone, and Cyclones II and III in series.

## Standard Air Flow

A critical issue affecting the regulation of PM is whether to report PM concentrations in terms of mass per actual unit volume or mass per dry standard unit volume. Currently, all air quality measurements that are expressed as mass per unit volume (e.g. $\mu g/m^3$) other than the $PM_{10}$ and $PM_{2.5}$ standards are corrected to a reference temperature of $25^oC$, a reference pressure of 790 mm Hg, and a reference relative humidity of 0% (*CFR*, 2001c). Measurements of $PM_{10}$ and $PM_{2.5}$, for purposes of comparison to the standards, are to be reported based on actual ambient air volume measured at the actual ambient temperature, pressure, and relative humidity at the monitoring site during the sampling period.

Wedding (1985) reported that the flow rate through inertial impactor should be maintained at “local” temperatures and pressures to retain the separator’s cutpoint calibration. Wedding (1995) also stated that the use of mass flow controllers may significantly affect the separator’s flow velocity during large diurnal temperature changes, causing excessively cutpoint errors. Although there have been some reports on the effects of using mass flow controllers to maintain a dry standard volume of air pulled by the sampler, the majority of the literature focuses the issue of subsequently correcting the sampled aerosol volume to standard conditions by mathematically compensating for average meteorological conditions.

The literature suggests that health effect related issues are the primary reasons for reporting PM mass concentrations in terms of actual unit volume. According to USEPA (1996a), “… the rationale for aerosol sampling was to mimic respiratory penetration (occurring at local conditions). A correction after the fact may not be appropriate”. Recent health effects studies have been conducted in cool and warm climates, and in cities at high altitudes (e.g. Denver) as well as near sea level (e.g. Philadelphia). Results from these studies have shown no evidence that the risks associated with PM exposures are affected by variations in altitude. These reports further suggest that adjusting temperature and pressure to dry standard conditions would not significantly change the reported concentration and would be below the detection limits of epidemiological studies. Although the delivery dose of PM might be expected to increase at extreme altitudes, for those not acclimatized to such locations, dosimetric studies have provided no clear support justifying any PM concentration adjustment to standard conditions.

## Particle Size Distributions

The distribution of particle mass with respect to particle size is perhaps the most important physical parameter governing particle behavior (USEPA, 1996a). Atmospheric deposition rates of particles, and therefore their residence time in the atmosphere, are strong functions of particle size. Particle deposition patterns in the human respiratory system are also governed by particle size (USEPA, 2001a). Particles that exist in the atmosphere as aerosols are airborne suspensions of finely dispersed solid or liquid particles. The diameters of atmospheric particles span five orders of magnitude, ranging from 1 nm to 100 μm. Atmospheric aerosols present in natural and work environments are poly-disperse, meaning the constituent particles within an aerosol have a range of sizes that can be appropriately described in terms of a size distribution function or the characteristic parameters describing the function.

Hinds (1982) indicated that most aerosols in the ambient air are poly-disperse and that the lognormal distribution “is the most common distribution used for characterizing the particle sizes associated with the aerosol”. The use of a lognormal function to approximate aerosol size distributions was first introduced by Foizik (1950) and later expanded to a wide range of atmospheric data by Willeke and Whitby (1975) and Whitby and Sverdrup (1980). A lognormal distribution function may not always be the best fit in describing a particular particle size distribution, but the goodness of fit associated with a lognormal distribution is typically close to that associated with the best fit function. The utility of the lognormal distribution is another attribute in using the function to describe particle size distributions in that the function can be characterized by the mass median diameter (MMD) and the geometric standard deviation (GSD). For example, since the mass of a material is usually more relevant to its potential toxicity, the MMD and GSD are usually preferred in describing aerosols in inhalation toxicology research (USEPA, 1982a).

In order to avoid the complications associated with defining particle diameters because of the effects of particle shape, size, and density on the inertial properties of airborne particles, aerodynamic diameters have been defined and used to classify particles with common inertial properties (USEPA, 1982a). The aerodynamic diameter most generally used is the aerodynamic equivalent diameter (AED), defined by Hatch and Gross (1964) as the diameter of a unit density sphere having the same terminal settling velocity as a particle with a differing geometric particle size, shape, and density. Another parameter often used in the aerosol science to describe particle diameter is Stokes diameter. Stokes diameter describes particle size based on the aerodynamic drag force imparted on a particle when its velocity differs from that of the surrounding fluid. For a smooth, spherically shaped particle, Stokes diameter exactly equals the physical diameter of the particle. For an irregularly shaped particle, Stokes diameter is the diameter of an equivalent sphere that would have the same aerodynamic resistance (i.e. particles of equal density and equal Stokes diameter have the same settling velocity). Fuchs (1964, 1989), Friedlander (1977), Reist (1984, 1993), Hinds (1982, 1999), Willeke and Baron (1993), and Seinfeld and Pnadis (1998) provide additional information on particle diameter definitions and mathematical relationships.

There are various methods or techniques currently used to determine the particle size distribution characteristics of PM, including but not limited to: aerodynamic separation (i.e. impactors and cyclones), microscopy, laser diffraction, time of flight, and electrical sensing zone. Aerodynamic separation methods are generally less expensive and simpler to use than the other methods; however, this method does not provide a distinct classification by size (USEPA, 1996a). Aerodynamic separation methods provide a limited number of size fractions, yielding a discontinuous function of particle size versus mass. Light microscopy has been used for determining particle size information regarding the morphology of microscopic features (Crutcher, 1982). The practical resolution of light microscopy is typically limited to 1 to 2 μm (Meyer-Arendt, 1972). Scanning microscopy provides qualitative results because of the limited number of particles counted per sample. The evolution of computer technology, pattern recognition algorithms, has improved the quantitative source apportionment of scanning microscopy (Bruynseels et al., 1988; Hopke and Casuccio, 1991). Laser diffraction techniques pass a jet of aerosol through an optical system where light is scattered from individual particles and detected by a photo-detector array. Discrete signals are counted and sorted by intensity, based on a refractive index selected by the user. The time of flight method, which is used to determine the particles velocity, accelerates the aerosol through a nozzle and past two laser beams. The particle velocity is related to the particle density and drag force, and the instrument is calibrated such that the aerodynamic particle size is known (Miller and Lines, 1998). The electrical sensing zone method pulls an aerosol sample, dispersed in an electrolyte solution, through an aperture tube and past a set of electrodes. The electrodes measure the increase in impedance as the particle passes through the system. This increase in impedance is proportional to the volume of electrolyte displaced by the particle (Beckman Coulter, 2000). There are advantages and disadvantages associated with each of these methods and unfortunately, there is no single agreed upon method of determining the particle size distribution characteristics of PM.

Particle size characteristics of PM associated with paved and unpaved roads, agricultural soil, sand and gravel, and alkaline lakebed sediments determined in a laboratory resuspension study by Chow et al. (1994), are listed in Table 3. Particle size fractions for road and soil dust, construction dust, agricultural burning, residential wood combustion, diesel truck exhaust, and crude oil combustion as determined by Houck et al. (1989, 1990) are also listed in Table 3. The data listed in Table 3 illustrates the particle size differences associated with PM emitted by differing sources. In general, the majority of particle mass associated sources of combustion corresponds to particle diameters less than 2.5 μm. Kleeman et al. (1999) reported that the particle sizes associated with the combustion of wood in fireplaces were predominately less than 1.0 μm. Radke et al. (1991) reported that approximately 70% of PM mass from biomass burning was associated with particle diameters less than 3.5 μm AED. The diameter of particles produced in the atmosphere by photochemical processes range in diameter from

0.003 to 2 μm (USEPA, 1996a). Particle diameters associated with combustion-generated particles, such as those from power generation, automobiles, and tobacco smoke can be as small as 0.003 μm and as large as 1 μm. Particle diameters of fly ash produced by coal combustion can range from 0.1 to 50 μm and particle diameters associated with windblown dust, pollens, plant fragments, and cement dusts are generally above 2 μm in diameter. Cowherd (1974) estimated that the $PM_{10}$ and $PM_{2.5}$ fraction of total dust emissions from agricultural tilling was 21 and 10%, respectively. Particulate matter characteristics for other defined modes, urban sources, agricultural sources, and miscellaneous sources (i.e. used in sampler evaluation studies) are listed in Table 4.

**Table 3.** Particulate matter size fraction estimates for various sources.

| | PM Percent (%) of TSP | | | |
|---|---|---|---|---|
| Source | < 1.0 μm | < 2.5 μm | < 10 μm | Reference |
| Road and Soil Dust | 4.5 | 10.7 | 52.3 | Houck et al. (1989, 1990) |
| Paved Road Dust | 4.0 | 10.0 | 48.0 | Chow et al. (1994) |
| Unpaved Road Dust | 4.0 | 9.0 | 56.0 | Chow et al. (1994) |
| Agricultural Soil | 4.0 | 12.0 | 56.0 | Chow et al. (1994) |
| Soil/Gravel | 6.0 | 15.0 | 35.0 | Chow et al. (1994) |
| Alkaline Lake Bed | 7.0 | 13.0 | 52.0 | Chow et al. (1994) |
| Construction Dust | 4.6 | 5.8 | 34.9 | Houck et al. (1989, 1990) |
| Agricultural Burning | 81.6 | 82.7 | 92.8 | Houck et al. (1989, 1990) |
| Residential Wood Combustion | 92.4 | 93.1 | 95.8 | Houck et al. (1989, 1990) |
| Diesel Truck Exhaust | 91.8 | 92.3 | 96.2 | Houck et al. (1989, 1990) |
| Crude Oil Combustion | 87.4 | 97.4 | 99.2 | Houck et al. (1989, 1990) |

**Table 4.** Characteristics of various types of particulate matter.

| Source | MMD (μm) | GSD | Particle Density (g/cm$^3$) | Reference |
|---|---|---|---|---|
| **Modes** | | | | |
| Nuclei | 0.05 – 0.07 | 1.8 | NR | USEPA (1996a) |
| Accumulation | 0.3 – 0.7 | 1.8 | NR | USEPA (1996a) |
| Coarse | 6.0 – 20.0 | 2.4 | NR | USEPA (1996a) |
| **Urban** | | | | |
| Urban Dust | 5.7 | 2.25 | NR | USEPA (1996a) |
| Wood Burning (Hardwood, Softwood, and Synthetic Logs) | 0.17 | NR | NR | Dasch (1982) |
| "In Traffic" | 4.6 | 1.49 | NR | Wilson and Suh (1997) |
| **Agricultural** | | | | |
| Rice | 21.75 | NR | NR | Plemons (1981) |
| Rice | 12.10 | 2.24 | 1.46 | Parnell et al. (1986) |
| Corn | 19.57 | NR | NR | Plemons (1981) |
| Corn | 13.70 | NR | NR | Wade (1979) |
| Corn | 13.60 | 1.80 | 1.50 | Parnell et al. (1986) |
| Soybeans | 25.17 | NR | NR | Plemons (1981) |
| Soybeans | 30.00 | NR | NR | Martin (1981) |
| Soybeans | 15.50 | NR | NR | Wade (1979) |
| Soybeans | 14.80 | 1.87 | 1.69 | Parnell et al. (1986) |
| Wheat | 32.97 | NR | NR | Plemons (1981) |
| Wheat | 14.70 | 2.08 | 1.48 | Parnell et al. (1986) |
| Sorghum | 36.92 | NR | NR | Plemons (1981) |
| Sorghum | 15.70 | 2.16 | 1.43 | Parnell et al. (1986) |
| Cotton Gin (Combined Streams) | 20 - 23 | 1.82 – 2.00 | 1.8 - 2.0 | Wang (2000) |
| Cotton Lint Fibers | 12.94 | 2.25 | NR | Parnell and Adams (1979) |
| Cattle Feedlot (Downwind) | 14.2 | 2.25 | 1.71 | Sweeten et al. (1989) |
| Swine Finishing House (Aerial) | 14.3 | 2.02 | NR | Barber et al. (1991) |
| Swine Finishing House (Settled) | 18.4 | 1.99 | NR | Barber et al. (1991) |
| Swine Production Facility | 17.97 | NR | NR | Barber et al. (1991) |
| Poultry Production Facility | 24.0 – 26.7 | 1.6 | NR | Redwine and Lacey (2001) |
| Typical Soil | 25 | 2.0 | 2.5 | Pargmann et al. (2000) |
| **Miscellaneous** | | | | |
| Arizona Road Dust | 6.0 | 3.0 | NR | Chen (1993) |
| Aloxite F1200 | 6.0 | 1.4 | NR | Mark et al. (1985) |
| Alumina | 8.1 | 1.51 | 3.91 | Pargmann (2001) |
| Aluminum Oxide | 8.4 | 1.4 | 3.9 | Wang et al. (2003) |
| Cornstarch | 19 | 1.4 | 1.5 | Wang et al. (2003) |
| Fly Ash | 12 | 1.7 | 2.7 | Wang et al. (2003) |

NR – Data not reported in the reference.

Columbus and Hughs (1993) conducted a cotton gin stack sampling study focused on the unloading and first stage lint cleaning exhausts. Columbus and Hughs (1993) used modified high volume samplers in conjunction with poly-web filter media. Test cottons were produced in various states consisting of various varieties. The MMDs and GSDs reported by Columbus and Hughs (1993) are listed in Table 5.

**Table 5.** MMDs and GSDs associated with the unloading and first stage lint cleaner exhausts for various cotton varieties from various states of origin (Columbus and Hughs, 1993).

| | | | Unloading | | Lint Cleaner | |
|---|---|---|---|---|---|---|
| State of Origin | Soil Type | Variety | MMD (μm) | GSD | MMD (μm) | GSD |
| AL | Decatur Silt Loam | DPL 50 | 5.84 | 2.25 | 10.1 | 2.41 |
| AR | Herbet Silt Loam | DPL20 | 5.27 | 2.20 | 8.2 | 2.26 |
| CA | Dundee Silt Loam | GC-510 | 3.87 | 2.00 | 12.5 | 2.17 |
| MO | Tiptonville | DES 119 | 5.76 | 2.32 | 8.6 | 2.28 |
| MS | Sandy Loam | Delcott 344 | 5.48 | 2.41 | 7.5 | 2.27 |
| NM | Clay Loam | Acala 1517-88 | 3.69 | 1.96 | 10.4 | 2.07 |
| NM Pima | Clay Loam | Pima | 4.11 | 2.00 | 9.5 | 1.98 |
| OK | Clay Loam | DPL90 | 5.40 | 2.35 | 6.0 | 2.19 |
| SC | Sandy Loam | DPL 5690 | 5.52 | 2.28 | 8.0 | 2.52 |
| TN | Colino Silt Loam | DPL50 | 4.96 | 2.28 | 7.3 | 2.24 |
| TX | Acuff Loam | Paymaster HS26 | 4.18 | 1.91 | 7.6 | 2.03 |

Hughs and Wakelyn (1996, 1997) acquired test filters from SAPRA required stack sampling conducted in 1994 at a cotton gin in New Mexico and another gin in California. Method 5 stack sampling was conducted at both gins on various exhausts. In addition, Method 501 was used at the California gin. Hughs and Wakelyn (1996, 1997) used the glass fiber filters obtained from the sampling tests to determine the fraction of $PM_{10}$ and $PM_{2.5}$ associated with the various exhausts using Coulter Counter analysis. The data reported by Hughs and Wakelyn (1996, 1997) are listed in Table 6. Hughs and Wakelyn (1996, 1997) reported differences between the two gins and noted that the size fractions determined by Method 501 were considerably lower than the results determined by the Coulter Counter method.

**Table 6.** PM fractions associated with various exhausts of a New Mexico and California cotton gin (Hughs and Wakelyn, 1996, 1997).

| New Mexico Gin | | | California Gin | | | |
|---|---|---|---|---|---|---|
| | Coulter Counter | | | Coulter Counter | | Method 501 |
| Exhaust | $PM_{2.5}$ (%) | $PM_{10}$ (%) | Exhaust | $PM_{2.5}$ (%) | $PM_{10}$ (%) | $PM_{10}$ (%) |
| Unloading | 2.1 | 68.9 | Unloading & 1st Dryer | 1.5 | 79.0 | 39.5 |
| 1st Hot Air Cleaner | 2.4 | 71.0 | Remaining Seed Cotton Cleaning | 0.7 | 72.9 | 27.5 |
| 2nd Hot Air Cleaner | 2.5 | 61.8 | Lint Cleaner Trash | 0.4 | 54.6 | 41.1 |
| Incline over Distributor | 2.2 | 70.9 | Battery Condenser | 0.6 | 59.5 | 41.6 |
| Motes | 2.2 | 74.5 | Motes Trash | 0.6 | 71.8 | 38.5 |

## Cotton Gin Emissions

According to 40 CFR, Part 60, Subpart DD (*CFR*, 2002) no grain elevator emission point (except for the grain dryer) should exceed 0.023 g/dscm (0.01 gr/dscf) TSP, as determined by EPA Method 5. The San Joaquin Valley Unified Air Pollution Control District has imposed a similar limitation on cotton gin exhausts through Rule 4201, which limits TSP emission concentrations from cotton gin exhausts to 0.23 g/dscm (0.1 gr/dscf). On October 16, 1995, the EPA issued a guidance memorandum clarifying the applicability of Title V in that source measurement of PM should be based on $PM_{10}$ and not TSP (Wegman, 1995). However, some SAPRA, such as those in the state of California, continue to regulate TSP emission concentrations at the stack.

The cotton ginning industry trend is toward fewer gins with higher processing capacities. In 1979, there were 2,332 active gins in the United States producing 14,161,000 bales of cotton (USEPA, 1995). By the 1990/1991 season, the number of cotton gins in the United States had dropped to 1,533 and production had increased to about 15,038,000 bales. According to the USEPA (1995), the PM emissions emitted by cotton gins are a function of the type of gin, geographic region, type of cotton, harvest method, trash content, climate, production rate, and type and number of controls used by the facility.

Holt et al. (2000) determined the quantity of trash produced from various cotton gin process streams for two stripper harvested cotton varieties (Paymaster HS26 and HS 200) for both field and non-field cleaned conditions. Results are shown in Table 7. Because of the limited quantity of material captured from some of the process streams, Holt et al. (2000) combined samples from similar sources. Holt et al. (2000) determined the percent of total trash associated with the combined samples (results shown in Table 8) and also conducted a sieve analysis on the combined samples (results shown in Table 9).

**Table 7.** Average weight of cotton gin trash generated from various ginning systems for HS-26 and HS-200 stripper varieties when field and non-field cleaned (Holt et al., 2000).

| System | HS-26 Field Cleaned (kg/bale) | HS-26 Non-Field Cleaned (kg/bale) | HS-200 Field Cleaned (kg/bale) | HS-200 Non-Field Cleaned (kg/bale) |
|---|---|---|---|---|
| Unloading | 15.30 | 22.91 | 11.11 | 18.14 |
| No. 6 Separator | 0.94 | 0.68 | 0.04 | 0.49 |
| Overflow | 0.28 | 0.07 | 0.00 | 0.00 |
| Feeder | 3.18 | 7.71 | 1.13 | 4.54 |
| Gin Stand | 4.46 | 1.29 | 0.82 | 1.20 |
| 1st Incline | 21.02 | 33.57 | 23.59 | 29.71 |
| 2nd Incline | 6.80 | 9.98 | 5.67 | 8.62 |
| 1st Extractor | 59.00 | 188.95 | 49.22 | 176.45 |
| 2nd Extractor | 15.13 | 40.37 | 8.16 | 26.08 |
| 1st Lint Cleaner | 11.40 | 13.04 | 11.26 | 12.14 |
| 2nd Lint Cleaner | 2.56 | 3.00 | 2.58 | 3.03 |
| Total | 140.04 | 321.57 | 113.58 | 280.40 |

**Table 8.** Percent of cotton trash produced by equipment category for HS-26 and HS-200 stripper varieties when field and non-field cleaned (Holt et al., 2000).

| Equipment Category | HS-26 Field Cleaned (% of total) | HS-26 Non-Field Cleaned (% of total) | HS-200 Field Cleaned (% of total) | HS-200 Non-Field Cleaned (% of total) |
|---|---|---|---|---|
| Unloading System | 10.9 | 7.1 | 9.8 | 6.5 |
| Feeder & Gin Stand | 6.3 | 3.0 | 1.8 | 2.2 |
| Inclines | 19.9 | 13.6 | 25.8 | 13.7 |
| Extractors | 52.9 | 71.3 | 50.5 | 72.2 |
| Lint Cleaners | 10.0 | 5.0 | 12.1 | 5.4 |

**Table 9.** Sieve analysis (% by weight) of stripper, field cleaned and non-field cleaned, gin trash processed by various ginning systems (Holt et al., 2000).

| | Sieve Size (mm) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Source | Pan | 0.08 | 0.18 | 0.71 | 1.40 | 8.00 | 9.50 | 16.00 | 19.00 | 22.40 |
| *Field Cleaned* | | | | | | | | | | |
| Unloading System | 16.79 | 9.72 | 22.10 | 8.26 | 9.04 | 0.43 | 2.06 | 0.63 | 2.32 | 30.03 |
| Feeder & Gin Stand | 2.35 | 1.56 | 9.00 | 8.40 | 38.97 | 2.74 | 11.00 | 1.98 | 4.54 | 19.60 |
| Incline Cleaners | 7.73 | 8.63 | 25.90 | 13.62 | 12.88 | 0.64 | 2.11 | 2.20 | 6.66 | 19.46 |
| Extractors | 0.18 | 0.19 | 0.74 | 2.32 | 16.97 | 6.04 | 49.15 | 10.14 | 8.54 | 5.08 |
| Lint Cleaners | 0.17 | 0.39 | 2.49 | 0.86 | 1.04 | 0.00 | 0.12 | 0.00 | 0.05 | 94.99 |
| *Non-Field Cleaned* | | | | | | | | | | |
| Unloading System | 15.57 | 7.85 | 16.18 | 6.26 | 14.32 | 1.85 | 16.47 | 5.81 | 4.19 | 4.80 |
| Feeder & Gin Stand | 0.96 | 0.82 | 6.26 | 6.35 | 33.98 | 3.75 | 18.61 | 2.18 | 5.36 | 21.13 |
| Incline Cleaners | 10.20 | 8.66 | 26.17 | 14.05 | 16.81 | 0.71 | 4.23 | 2.43 | 12.89 | 3.34 |
| Extractors | 0.21 | 0.24 | 0.27 | 0.85 | 8.69 | 2.39 | 43.56 | 29.10 | 10.91 | 3.92 |
| Lint Cleaners | 0.12 | 0.36 | 1.67 | 0.79 | 1.23 | 0.00 | 0.00 | 0.00 | 0.02 | 96.13 |

Rawlings and Reznik (1978) defined a “representative cotton gin” as having an annual production of 4,200 bales (217 kg/bale) per year, operating capacity of 6.8 bales/h, and a operating schedule of 10 h/day, 6 days/week, and 600 h/year. During the 1976 crop year, approximately 10.58 million bales of cotton were ginned in 18 southern and western states (Department of Commerce, 1976). Rawlings and Reznik (1978) reported that on a national basis, emissions from cotton gins in 1976 represented 0.04% of the total annual TSP emissions. Rawling and Reznik (1978) compiled average emission factor data for stripper, picker, and a defined representative cotton gin, which are shown in Table 10.

Parnell and Baker (1973) reported that the trash content in seed cotton increased with extended harvest dates and that emission factors increased with increased trash content. Parnell and Baker (1973) also reported that emission concentrations decreased with increases in ginning rate (i.e. slower ginning rates resulted in higher emission concentrations).

**Table 10.** Average emission factors for stripper gins processing early season, midseason, late season, and extremely dirty cotton (Parnell and Baker, 1973), picker gins (Rawlings and Reznik, 1978), and a representative gin (Rawlings and Reznik, 1978).

| | Average Emission Factor, kg/bale (lb/bale) | | | | | |
|---|---|---|---|---|---|---|
| System | Early Season (Stripper) | Midseason (Stripper) | Late Season (Stripper) | Extremely Dirty (Stripper) | Picker | Representative |
| Unloading | 0.099 | 0.076 | 0.144 | 0.748 | 0.056 | 0.066 |
| | (0.218) | (0.168) | (0.317) | (1.650) | (0.124) | (0.146) |
| 1st Dryer and Cleaner | 0.025 | 0.086 | 0.082 | 0.198 | 0.035 | 0.056 |
| | (0.056) | (0.190) | (0.180) | (0.437) | (0.076) | (0.124) |
| 2nd Dryer and Cleaner | 0.014 | 0.033 | 0.035 | 0.062 | 0.040 | 0.035 |
| | (0.030) | (0.072) | (0.078) | (0.136) | (0.089) | (0.077) |
| Extractors | NR | NR | NR | NR | 0.011 | 0.006 |
| | | | | | (0.025) | (0.013) |
| Overflow & Distributor | 0.036 | 0.017 | 0.023 | 0.038 | 0.041 | 0.054 |
| | (0.080) | (0.038) | (0.050) | (0.084) | (0.091) | (0.118) |
| 1st Lint Cleaner | 0.160 | 0.266 | 0.481 | 0.515 | 0.142 | 0.205 |
| | (0.352) | (0.587) | (1.060) | (1.136) | (0.314) | (0.452) |
| 2nd Lint Cleaner | 0.023 | 0.036 | 0.041 | NR | 0.084 | 0.060 |
| | (0.050) | (0.079) | (0.090) | | (0.186) | (0.133) |
| Mote | 0.038 | 0.054 | 0.060 | 0.010 | 0.060 | 0.057 |
| | (0.084) | (0.118) | (0.133) | (0.220) | (0.133) | (0.126) |
| Battery Condenser | 0.034 | 0.034 | 0.031 | 0.043 | 0.113 | 0.074 |
| | (0.074) | (0.074) | (0.068) | (0.095) | (0.249) | (0.162) |
| Master Trash | 0.019 | 0.099 | 0.075 | 0.122 | 0.054 | 0.072 |
| | (0.042) | (0.219) | (0.166) | (0.270) | (0.120) | (0.158) |
| Total | 0.447 | 0.701 | 0.972 | 1.827 | 0.639 | 0.685 |
| | (0.986) | (1.545) | (2.142) | (4.028) | (1.408) | (1.509) |

NR – Not Reported

The 1996 EPA AP-42 emission factors are based on results from emission tests conducted at 10 gins (nine in California and one in Tennessee) (USEPA, 1996b). The 1996 EPA AP-42 TSP emission factors were determined by Method 5 or CARB Method 5 and the $PM_{10}$ emission factors were determined by CARB 501 (cascade impactor). The 1996 EPA AP-42 emission factors are listed in Table 11. The California Cotton Ginners Association also published a list of $PM_{10}$ emission factors for saw-type and roller-type gins equipped with various abatement devices, shown in Table 12. The California Cotton Ginners Association $PM_{10}$ emission factors were determined through the use of CARB Method 501 (tests conducted prior to 1996) and EPA's Method 201A (tests conducted after 1996).

**Table 11.** 1996 EPA AP-42 cotton gin emission factors (USEPA, 1996b).

| Process Stream | TSP, kg/bale (lb/bale) | $PM_{10}$, kg/bale (lb/bale) |
|---|---|---|
| Unloading | 0.132 (0.29) | 0.054 (0.12) |
| Module Feeder | NR | NR |
| 1st Stage Seed Cotton Cleaning | 0.163 (0.36) | 0.054 (0.12) |
| 2nd Stage Seed Cotton Cleaning | 0.109 (0.24) | 0.042 (0.093) |
| 3rd Stage Seed Cotton Cleaning | 0.043 (0.095) | 0.015 (0.033) |
| Distributor | 0.032 (0.071) | 0.012 (0.026) |
| Overflow | NR | NR |
| Trash | 0.245 (0.54) | 0.034 (0.074) |
| Cyclone Robber | 0.082 (0.18) | 0.024 (0.052) |
| Mote | 0.127 (0.28) | 0.059 (0.13) |
| Mote Trash | 0.035 (0.077) | 0.010 (0.021) |
| 1st Stage Lint Cleaning | | |
| (Covered Condenser Drum) | | |
| (Cyclone) | *1st and 2nd Stages Combined* | |
| 2nd Stage Lint Cleaning | 0.499 (1.1) | NR |
| (Covered Condenser Drum) | 0.263 (0.58) | 0.109 (0.24 |
| (Cyclone) | | |
| 3rd Stage Lint Cleaning | | |
| (Covered Condenser Drum) | NR | NR |
| (Cyclone) | NR | NR |
| Battery Condenser | | |
| (Covered Condenser Drum) | 0.077 (0.17) | NR |
| (Cyclone) | 0.018 (0.039) | 0.006 (0.014) |

NR – Not Reported

**Table 12.** Average $PM_{10}$ emission factors for saw and roller gins with various controls (California Cotton Ginners Association, 1997).

| System | Average Emissions, kg $PM_{10}$/bale (lb $PM_{10}$/bale)<br>Saw Gin with 2D-2D controls | <br>Saw Gin with 1D-3D controls | <br>Roller Gin with 1D-3D controls | <br>Saw Gin with Screen Baskets |
|---|---|---|---|---|
| Unloading | 0.095 (0.21) | 0.054 (0.12) | 0.136 (0.30) | NR |
| #1 Pre-Cleaning | 0.132 (0.29) | 0.041 (0.09) | 0.141 (0.31) | NR |
| #2 Pre-Cleaning | 0.095 (0.21) | 0.027 (0.06) | 0.064 (0.14) | NR |
| #3 Pre-Cleaning | 0.054 (0.12) | 0.059 (0.13) | 0.068 (0.15) | NR |
| Overflow | 0.018 (0.04) | 0.014 (0.03) | 0.014 (0.03) | NR |
| Gin Stand/Feeder Trash | 0.018 (0.04) | 0.032 (0.07) | 0.032 (0.07) | NR |
| #1 Lint Cleaning [1] | NR | 0.045 (0.10) | 0.036 (0.08) | 0.218 (0.48) |
| #2 Lint Cleaning [1] | NR | 0.014 (0.03) | NR | 0.136 (0.30) |
| Lint Cleaning [2] | 0.331 (0.73) | 0.045 (0.10) | 0.045 (0.10) | 0.354 (0.78) |
| Lint Trash/Robber | 0.109 (0.24) | 0.023 (0.05) | 0.009 (0.02) | NR |
| Battery Condenser | NR | 0.018 (0.04) | 0.041 (0.09) | 0.077 (0.17) |
| Motes | 0.113 (0.25) | 0.032 (0.07) | NR | NR |
| Mote Cleaner Trash | 0.009 (0.02) | 0.009 (0.02) | NR | NR |
| Stockpiler | 0.041 (0.09) | 0.027 (0.06) | 0.027 (0.06) | NR |
| Total [3] | 1.016 (2.24) | 0.381 (0.84) | 0.576 (1.27) | 0.431 (0.95) |

Note: no average emission were reported for roller gins with 2D-2D cyclones or screen baskets.
[1] Use when lint cleaner condenser fan is pulling from a single stage of condensers
[2] Use when lint cleaner condenser fan is pulling from both 1st and 2nd stage condensers
[3] Assumes total "lint cleaning" emission factor instead of individual stages.

## Cotton Gin Abatement Technologies

Most states, including Texas have phased out, or are phasing out, the use of "grandfathered" clauses and are requiring cotton gins to implement Best Available Control Technologies (BACT). Under most grandfathered clauses, cotton gins were not required to modify their existing air pollution abatement technologies as long as no changes (e.g., gin machinery upgrades, fan upgrades, and production rate increases) were made to the gin that would affect the gins emission output. BACT is defined as an emission limitation based on the maximum degree of emission reduction (with consideration given to the technical practicability and economic reasonableness of reducing or eliminating emissions from the facilities exhausts) achievable through application of production processes and available methods, systems, and techniques

(TACB, 1992). However, BACT does not permit emissions in excess of those allowed under any applicable CAA provisions. Several SAPRAs define BACT for cotton gins as high efficiency cyclones (1D-3D or 2D-2D) on all centrifugal fan exhausts and covered condenser drums with 70-100 fine-mesh screens on all axial fan exhausts (e.g. lint cleaners and battery condensers).

Cyclones are predominately used in controlling cotton gin PM emissions; however, other technologies are used or have been explored in controlling cotton gin PM emissions, such as covered condenser drums, gravity settling chambers, baffle-type pre-separators, series cyclones, rotary drum filters, precipitators, scrubbers, and bagfilters. Two primary reasons for the wide use of cyclone technology are the relatively low capital costs and relatively low maintenance requirements. Some of the cyclone designs currently used in the cotton ginning industry include: 1D-3D (with a traditional 1D-3D inlet, inverted 1D-3D inlet, or a 2D-2D inlet), 2D-2D, or 1D-2D. Cyclone collection efficiencies are reported to vary as a function of the particle size of the material being separated from the air and by cyclone design (USEPA, 1998). According to USEPA (1998), cyclone efficiency generally increases with: 1) particle size and/or density of the material being separated from the air; 2) inlet air velocity; 3) cyclone body length; 4) number of gas revolutions in the cyclone; 5) ratio of cyclone body diameter to exit diameter; 6) dust loading; and 7) smoothness of the cyclone inner wall. According to USEPA (1998), cyclone efficiency is reported to decrease with increases in: 1) gas viscosity; 2) body diameter; 3) exit diameter; 4) inlet area; and 5) air density.

Early cyclones used in the cotton ginning industry were large-diameter, low-velocity devices designed primarily for the collection of large trash. During the 1960’s, the high-efficiency, small-diameter cyclone, commonly referred to as the 2D-2D design, was developed for the cotton ginning industry in an effort to reduce PM emissions (Harrell and Moore, 1962; Baker and Stedronsky, 1967). In the late 1970’s, Parnell and Davis (1979) introduced the 1D-3D cyclone design which was reported to have a higher collection efficiency, under fine dust loadings, than the 2D-2D cyclone design. EC/R Incorporated (1998) reported that single conventional cyclones could remove 10 μm particles with 85 - 90% efficiency, 5 μm particles with 75-85% efficiency, and 2.5 μm particles with 60 - 75% efficiency. Avant et al. (1976) reported the high efficiency cyclone could collect particles greater than 20 μm with 100% efficiency. EC/R Incorporated reported that single high efficiency cyclones could remove 5 μm particles with 90% efficiency.

High efficiency 1D-3D or 2D-2D cyclones are generally used on centrifugal fan exhausts. In the past, vane-axial fans were used for lint cleaner and battery condenser exhausts; however, the current trend is towards the use of centrifugal fans on these exhausts. Covered condenser drums are simply the condenser drum covered with 70-100 mesh screen wire or perforated metal (Columbus and Anthony, 1991). Covered condenser drums are among the least expensive controls available. Lint cleaner and battery condenser exhausts are associated with high air flow rates and high lint fiber. Columbus and Anthony (1991) reported that 25% of the material exiting a lint cleaner covered condenser drum was lint fiber. Covered condenser drums are estimated to be 50% efficient (Parnell et al., 1994).

As stated previously, the trend within the cotton industry is to replace covered condenser drums with cyclone technology. One critical issue associated with using 1D-3D or 2D-2D cyclones on lint cleaner or battery condenser exhausts is the increased pressure drop associated with adding the cyclones (i.e. the vane axial fans will have to be replaced with centrifugal fans if properly sized 1D-3D or 2D-2D cyclones are used on these exhausts). Milhalski et al. (1993) and Baker and Hughs (1996) reported "cycling lint" near the trash exit of 1D-3D and 2D-2D cyclones when used on high lint exhaust. Milhalski et al. (1993) reported significant increases in PM concentrations for 1D-3D and 2D-2D cyclones when processing high lint fiber material and attributed the increases to "cycling lint".

Baker et al. (1996) reported that properly sealed 1D-3D or 2D-2D cyclones retro-fitted with large expansion chambers (at the trash exit) would improve PM collection efficiency and would reduce the problems associated with "cycling lint". Simpson and Parnell (1995) introduced a new low-pressure cyclone, referred to as the 1D-2D cyclone.

The 1D-2D cyclone consumes roughly one-third of the energy required of a 1D-3D or 2D-2D cyclone (i.e. a 1D-2D cyclone can be installed on a lint cleaner exhaust without replacing the fan system). Tulles et al. (1997) and Flannigan et al. (1997) reported that PM concentrations were significantly reduced when using 1D-2D cyclones, as compared to 1D-3D or 2D-2D cyclones, on high lint fiber exhausts. Wang (2000) concluded that the 1D-3D cyclone design was the most efficient cyclone for exhausts processing fine dust and/or large trash (i.e., low lint fiber) and that the 1D-2D cyclone design was the most efficient cyclone for high lint fiber exhausts.

Secondary abatement technologies such as baffle-type pre-separators or series cyclones have been incorporated in some instances. EC/R Incorporated (1998) stated that baffle-type pre-separators utilize inertia in addition to gravity and have $PM_{10}$ collection efficiencies approaching 20%. However, Baker et al. (1996) reported that including a baffle-type pre-separator prior to a 1D-3D cyclone did not improve the cyclones efficiency. Gillum et al. (1982) reported that series cyclones could reduce emissions by approximately 50% when compared to a single cyclone, but the energy requirements for the system were more than doubled. Gillum and Hughs (1983) reported a 40% reduction in PM emissions when utilizing cyclones in series operated at inlet velocities lower than the recommended design velocities.

The next level of technology above cyclone separators is filtration (i.e. rotary drum filters or baghouses). According to Parnell (1990), filtration technologies are expected to reduce emission concentrations to 23 $mg/m^3$ (0.01 gr/dscf); whereas Parnell (1990) concluded that properly designed 1D3D and 2D2D cyclones could achieve emission concentrations of less than 70 $mg/m^3$ (0.03 gr/dscf) at loading rates as high as 9 $g/m^3$. Rotary drum filters, when installed at cotton gins, are preceded by cyclones. Yarlagadda (1995) reported that the efficiency of rotary drum filters range from 80-90%, with a loading rate of 3 $g/m^3$. According to Parnell (1990), the annual operating costs for a filtration system could be 5 to 10 times higher than that for cyclone technology, bringing into question the economic reasonableness associated with implementing such a system.

EPA recommend that states keep cost-effectiveness of control measures under \$11,023/metric ton (\$10,000/ton) of reduced emissions (USEPA, 1997). Flannigan (1997) estimated that the cost associated with covered condenser drums was approximately \$17.66 per $m^3$/min (\$0.50/cfm). Ramaiyer (1996) and Mayfield et al. (1996) reported that the cost of a baffle-type pre-separator was approximately \$17.66 per $m^3$/min (\$0.50/cfm), installed. Brinkley et al. (1992) estimated that the average for cost for a 2D-2D cyclone, 1D-3D cyclone, 2D-2D & 1D-3D series cyclones, and 2D-2D cyclone followed by a rotary drum filter was \$24.72 per $m^3$/min (\$0.70/cfm), \$32.49 per $m^3$/min (\$0.92/cfm), \$56.86 per $m^3$/min (\$1.61/cfm), and \$95.70 per $m^3$/min (\$2.71/cfm), respectively. Ramaiyer (1996) and Mayfield et al. (1996) estimated that the average cost for cyclone technology was and \$35.31 per $m^3$/min (\$1.00/cfm), including transitions and installation. Yarlagadda and Parnell (1994) estimated that the average cost associated with rotary drum filters was and \$88.29 per $m^3$/min (\$2.50/cfm). In addition to alternative abatement technology costs, some states require source sampling be conducted after the modification has been completed. Source sampling costs are \$3,000 to \$4,000 per emission point (California Cotton Ginners Association, 2000).

# METHODS AND PROCEDURES

The methods and procedures used are broken down by objective and further segregated by secondary topics. The main sections, corresponding to the objectives, are inherent sampler errors and cotton gin exhaust PSD estimates.

## Inherent Sampler Errors

The inherent sampler errors associated with EPA approved $PM_{10}$ and $PM_{2.5}$ ambient air samplers, EPA approved $PM_{10}$ stack samplers, and EPA approved methods of determining $PM_{coarse}$ and the ratio of $PM_{2.5}$ to $PM_{10}$ were determined through mathematical simulations. These simulations were limited to inherent errors associated with established tolerances for sampler performance characteristics, the interaction of particle size distribution (PSD) characteristics and sampler performance characteristics, and the potential errors associated with sampler performance characteristics varying beyond the established tolerances. The governing equations and parameters used in the simulations are discussed in the following sections: 1) particle size distributions, 2) sampler performance characteristics, 3) estimating sampler and true concentrations, and 4) estimating the relative differences between sampler and true concentrations.

### Particle Size Distributions

The distribution of particles with respect to size is perhaps the most important physical parameter governing their behavior. Aerosols containing only particles of a particular size are called monodisperse while those having a range or ranges of sizes are called polydisperse. Hinds (1982) indicated that most aerosols in the ambient air are polydisperse and that the lognormal distribution “is the most common distribution used for characterizing the particle sizes associated with the aerosol”. A lognormal distribution is a specific form of the size distribution function for which the population of particles follows a Gaussian distribution with respect to the natural log of the particle diameter, $d_p$. The significance of using a lognormal distribution is that the PSD can be described in terms of the mass median diameter (MMD) and the geometric standard deviation (GSD). The mathematical definition and manipulation of the lognormal distribution used herein was also described, in a similar fashion, by Hinds (1998) and Seinfeld and Pandis (1997). The lognormal mass density function is expressed as:

$$f(d_p, MMD, GSD) = \frac{1}{d_p \, ln(GSD)\sqrt{2\pi}} exp\left[\frac{-\left[ln(d_p) - ln(MMD)\right]^2}{2\left[ln(GSD)\right]^2}\right] \quad (1)$$

for poly-disperse particles, where the *GSD* is greater than 1.0. For mono-disperse particles (i.e. *GSD* is equal to 1.0), the mass density function is equal to 1.0 when $d_p$ is equal to the *MMD* and zero for all other $d_p$ values. Mono-disperse particles are commonly used in evaluating samplers in a laboratory setting. During the evaluation process, various mono-disperse particle sizes are commonly used. This range of particle sizes can be described as a uniform distribution assuming constant particle concentrations for each individual size. The uniform density function is expressed as:

$$f(d_p, n, R) = \begin{cases} \frac{1}{n} & if \quad 0 < d_p < R \quad and \quad n \geq 1 \\ 0 & if \quad d_p \geq R \quad and \quad d_p < 0 \end{cases} \quad (2)$$

where *n* is the number of mono-disperse particle sizes used and *R* is the largest mono-disperse particle size.

For a lognormal distribution, the fraction of the total particles, *df*, having diameters between $d_p$ and $d_p + dd_p$ is

$$df = f(d_p, MMD, GSD) dd_p \quad (3)$$

where $dd_p$ is a differential interval of particle size. The area under the density distribution curve is always

$$\int_0^\infty f\left(d_p, MMD, GSD\right)dd_p = 1.0 \tag{4}$$

This area can be estimated by the following discrete summation

$$\sum f_i = \sum_i (h_i \Delta d_i) = 1.0 \tag{5}$$

where $h_i \Delta d_i$ is equal to the fraction $f_i$ of particles in the size range $\Delta d_i$. The area under the density function may be estimated for particle sizes ranging from zero to infinity, as in equation 4, between given sizes $a$ and $b$, or it may be the small interval $dd_p$. The area under the density function curve between two sizes $a$ and $b$ equals the fraction of particles whose diameters fall within this interval, which can be expressed continuously as

$$f_{ab}\left(a, b, MMD, GSD\right) = \int_a^b f\left(d_p, MMD, GSD\right)dd_p \tag{6}$$

or discretely as

$$f_i = \frac{n_i}{N} = (h_i \Delta d_i) \tag{7}$$

where

$$N = \sum_i \left(h_i^{'} \Delta d_i\right) \tag{8}$$

and $N$ is used to standardize for sample size. When using the discrete summation, $\Delta d_i$ should be relatively small to minimize the error associated with this estimation method.

Size distributions can also be presented as a cumulative distribution function, *F(a,MMD,GSD)*, defined as

$$F(a, MMD, GSD) = \int_0^a f(d_p, MMD, GSD)dd_p \qquad (9)$$

where *F(a,MMD,GSD)* is the fraction of the particles having diameters less than *a*. The fraction of particles having diameters between sizes *a* and *b*, $f_{ab}(a,b,MMD,GSD)$, can be determined directly by subtracting the cumulative fraction for size *a* from that for size *b*, as shown in equation 10.

$$f_{ab}(a, b, MMD, GSD) = F(b, MMD, GSD) - F(a, MMD, GSD) \qquad (10)$$

The concentration of particles having diameters between sizes *a* and *b*, $C_{ab}(a,b,MMD,GSD)$, can be expressed as

$$C_{ab}(a, b, MMD, GSD) = C_T\left(F(b, MMD, GSD) - F(a, MMD, GSD)\right) \qquad (11)$$

where $C_T$ is the total concentration of PM.

For a lognormal distribution, the mode < median < mean. A lognormal density distribution defined by a MMD of 20 μm and a GSD of 3.0 is shown in Figure 10 to illustrate the differences between the mode, median, and mean of a lognormal distribution. Lognormal density distributions defined by a MMD of 10 μm and GSD of 1.1, 1.5, and 3.0 are shown in Figure 11 to illustrate how the lognormal distribution is effected by increases in GSD values. Typically, the x-axis of a lognormal distribution is displayed on a log scale; however, the x-axis in Figures 10 and 11 are not displayed on a log scale; in order to graphically show the effects MMD and GSD on lognormal PSD's. Three important observations should be noted for lognormal distributions: (1) the mode shifts significantly to the left as the GSD increases, (2) the median is not affected by the increase in GSD, and (3) the larger the GSD the more closely the lognormal distribution is to a uniform distribution.

The general mathematical simulations, using the PSD governing equations, utilized *MMD* values ranging from 1 to 40 µm in intervals of 1 µm and *GSD* values ranging from 1.3 to 2.5. A focus of the simulations will center around: 1) a *MMD* of 5.7 µm and a *GSD* of 2.25 (EPA defined PSD characteristics for urban dust); 2) a *MMD* of 10 µm and a *GSD* of 1.5; 3) a *MMD* of 10 µm and a *GSD* of 2.0; 4) a *MMD* of 20 µm and a *GSD* of 1.5 (similar to some agricultural dusts); and 5) a *MMD* of 20 µm and a *GSD* of 2.0 (similar to some agricultural dusts).

## Sampler Performance Characteristics

A sampler's performance is generally described by either a cumulative collection or penetration efficiency curve. The "sharpness of cut" of the sampler pre-separator or the "sharpness of slope" of the sampler penetration efficiency curve significantly impacts the accuracy of sampler measurements. Three terms are often used to describe the sharpness of the penetration curve and are frequently and inappropriately interchanged. These terms are ideal, true, and sampler. An ideal penetration curve corresponds to data provided in 40 CFR, Part 53 (USEPA, 2001b). A true penetration curve can be described as a step function. In other words, all particles less than or equal to the size of interest are captured on the filter and all particles greater than the particle size of interest are captured by the pre-separator. Sampler refers to the actual penetration curve associated with a particular sampler.

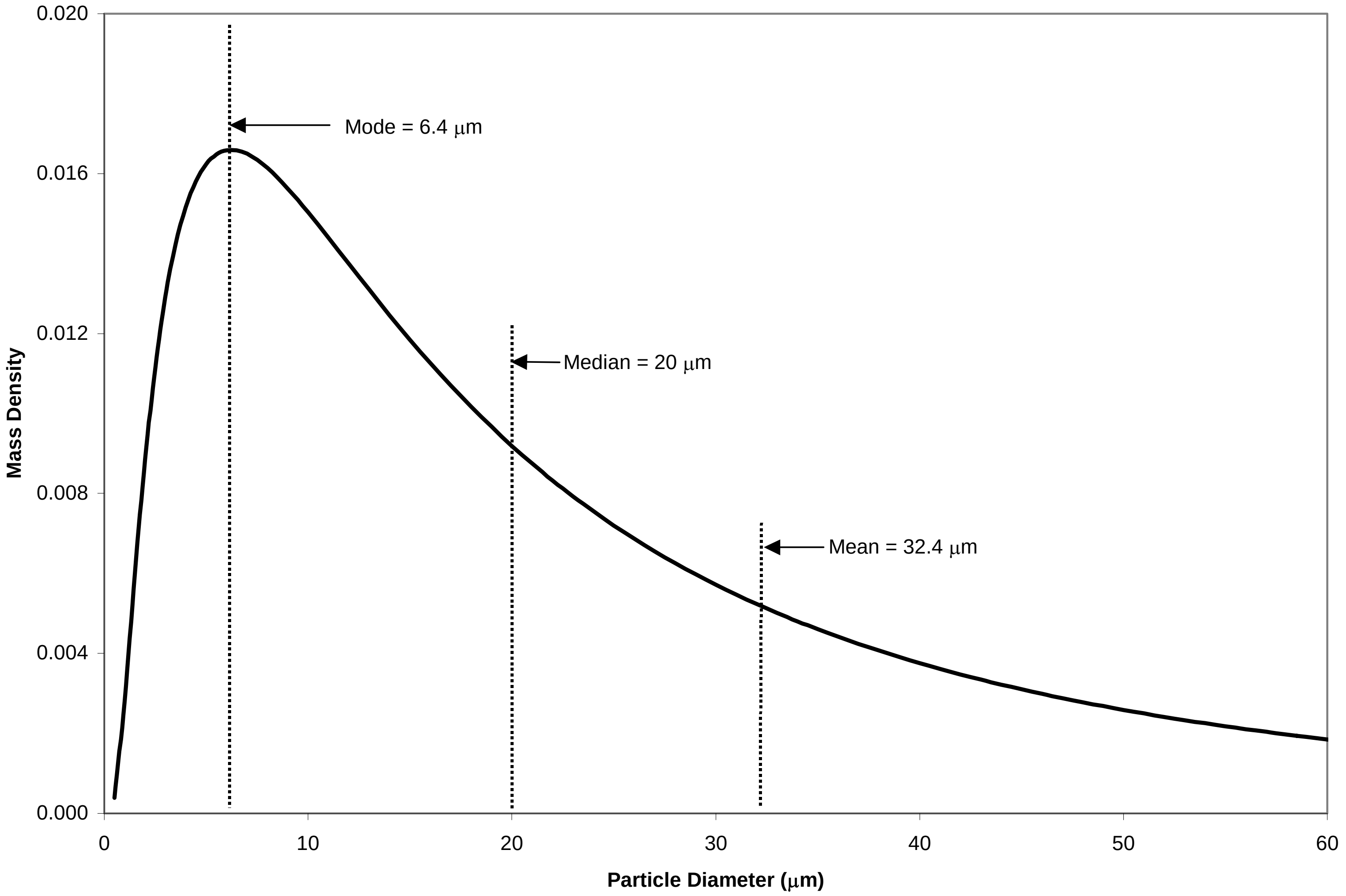


**Figure 10.** Lognormal particle size distribution defined by a MMD of 20 μm and a GSD of 3.0.

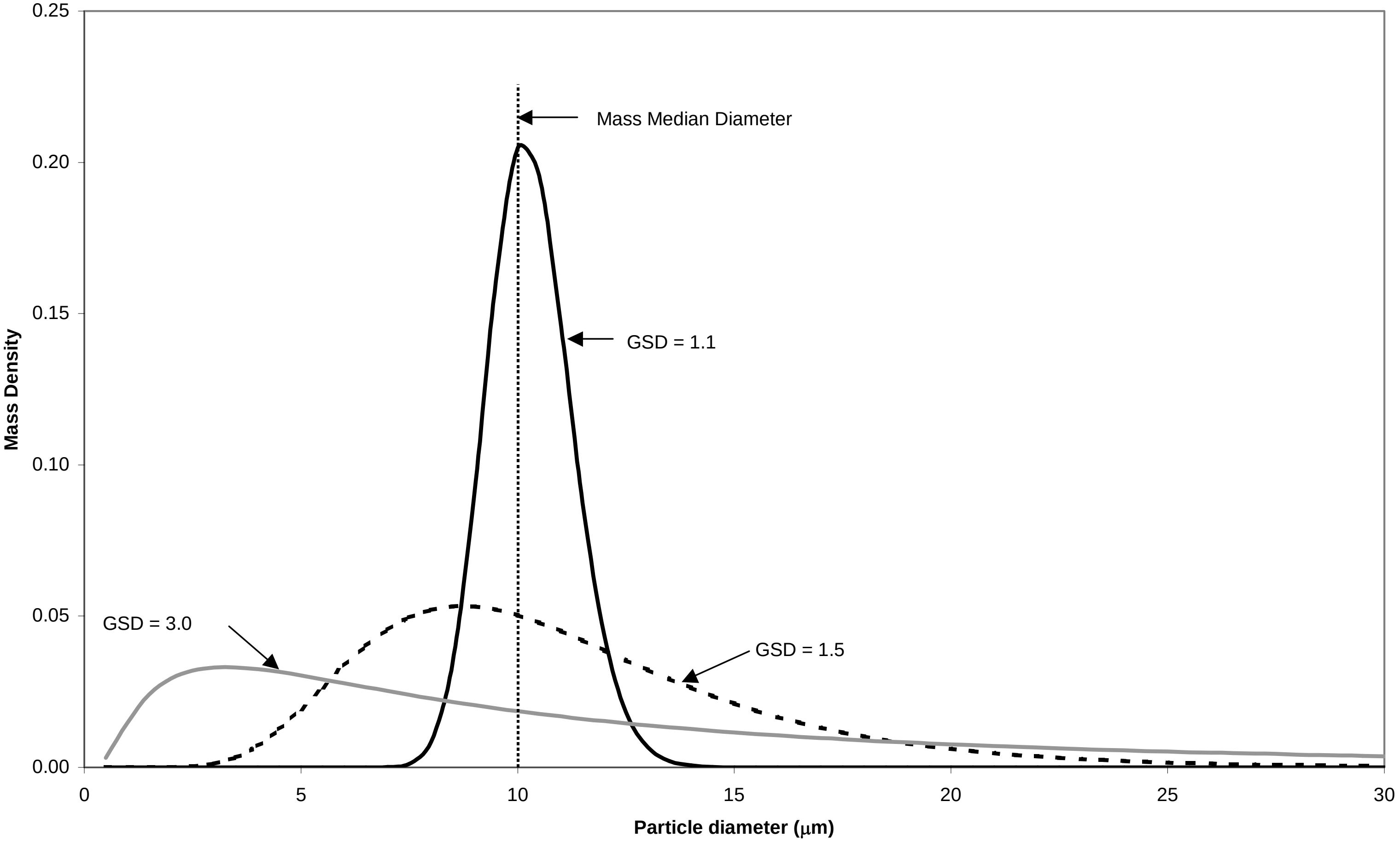


**Figure 11.** Lognormal particle size distributions described by a MMD of 10 μm and various GSDs.

A sampler penetration curve is defined by performance characteristics and based on these characteristics; a portion of PM less than the size of interest will not be collected on the filter (i.e. captured by the pre-separator) and a portion of the PM greater than the size of interest will be collected on the filter (i.e. should have been captured by the pre-separator). A common perception is that $PM_{10}$ and $PM_{2.5}$ sampler measured concentrations are true concentrations and that these concentrations relate to PM with particle sizes less than 10 and 2.5 μm, respectively; however, these measurement concentrations are actually based on the sampler performance characteristics.

A sampler's pre-separator collection efficiency curve is most commonly represented by a cumulative lognormal distribution and characterized by a $d_{50}$ (also referred to as cutpoint) and a slope. By definition, cutpoint is the particle size where 50% of the PM is captured by the pre-separator and 50% of the PM penetrates to the filter. Slope is defined as the ratio of particle sizes corresponding to cumulative collection efficiencies of 84.1% and 50% ($d_{84.1}/d_{50}$), 50% and 15.9% ($d_{50}/d_{15.9}$), or the square root of 84.1% and 15.9 % ($\sqrt{d_{84.1}/d_{15.9}}$). Collection efficiency curves are usually assumed as constant and independent of particle size. In other words, it is assumed that a significant loading of large particles does not affect the pre-separators collection efficiency for smaller particles. Therefore, concentration data used to generate a sampler's pre-separator collection efficiency curve is typically determined by conducting an array of tests over several mono-disperse particle sizes using known concentrations. The concentration data from each test is used to determine the collection efficiency, $\varepsilon_m$, associated with each particle size, using equation 12.

$$\varepsilon_m = \frac{C_{Pre-Separator}}{C_{test}} \tag{12}$$

In equation 12, $C_{Pre-Separator}$ is the concentration of particles captured by the pre-separator and $C_{test}$ is the concentration of particles used for the test. A smooth lognormal curve is fit to the calculated pre-separator collection efficiencies and the sampler performance characteristics ($d_{50}$ and *slope*) are determined from the fitted curve. The mathematical definition and manipulation of the lognormal collection efficiency curve used herein was also described, in a similar fashion, by Hinds (1998) and Seinfeld and Pandis (1997). The lognormal density distribution function for collection efficiency is defined as

$$\varepsilon_m\left(d_p, d_{50}, slope\right) = \left[\frac{1}{d_p\, ln(\,slope\,)\sqrt{2\pi}}\, exp\left[\frac{-\left[ln\left(d_p\right) - ln\left(d_{50}\right)\right]^2}{2(ln(slope))^2}\right]\right] \qquad (13)$$

Equation 13 applies to a sampler collection efficiency were the slope is greater than 1.0. An alternative equation is used to determine the true cut collection efficiency when the slope is equal to 1.0. Mathematical derivations for determining the cumulative distribution function for the collection efficiency can be achieved in the same manner as presented in the particle size distribution section.

The cumulative distribution function for the collection efficiency, $\psi(a,d_{50},slope)$, is defined by

$$\psi_m(a, d_{50}, slope) = \int_0^a \varepsilon_m(d_p, d_{50}, slope)\, dd_p \qquad (14)$$

where $\psi(a,d_{50},slope)$ gives the collection efficiency for particles having diameters less than *a*. The penetration efficiency, $P_m(a,d_{50},slope)$, is defined as

$$P_m(a, d_{50}, slope) = 1 - \psi_m(a, d_{50}, slope) \qquad (15)$$

Substituting equations 13 and 14 into equation 15 yields

$$P_m\left(a, d_{50}, slope\right) = 1 - \int_0^a \left[\frac{1}{d_p\, ln(\,slope\,)\sqrt{2\pi}}\, exp\left[\frac{-\left[ln\left(d_p\right) - ln\left(d_{50}\right)\right]^2}{2(ln(slope))^2}\right]\right] dd_p \qquad (16)$$

where $P_m(a,d_{50},slope)$ is the sampler penetration efficiency for particles having diameters less than $a$.

The true penetration curve is defined by a step function and defined as

$$P_v(a,d_{50},slope)=\begin{cases}1 & if\ a\le d_{50}\\ 0 & if\ a>d_{50}\end{cases} \tag{17}$$

Now that the penetration function has been defined, the sampler performance characteristics for the $PM_{10}$ and $PM_{2.5}$ samplers need to be defined in terms of $d_{50}$ and *slope*. The EPA essentially defines these parameters for the ambient air samplers in 40 CFR, Part 53 in the discussion of tests required for a candidate sampler to receive EPA approval. The $d_{50}$ for both the $PM_{10}$ and $PM_{2.5}$ ambient air samplers are explicitly stated in the EPA standards as 10.0 ± 0.5 μm AED and 2.5 ± 0.2 μm AED, respectively. No *slope* values for either sampler are listed in 40 CFR, Part 53 or any other current EPA standard; however, penetration data is presented 40 CFR, Part 53. Ideally, the penetration data could be fit to a cumulative lognormal distribution to determine the characteristic $d_{50}$ and *slope* for each of the samplers; however, it was found that no single cumulative lognormal curve adequately represented the data sets.

The $PM_{10}$ cumulative penetration data set produced a rough curve which appeared to have a larger *slope* for the particle sizes less than 10 μm AED than the *slope* for the particle sizes greater than 10 μm AED. Hinds (1982) suggested that the *slope* associated PM deposited in the thoracic region of the human respiratory system was 1.5 ± 0.1 and that this *slope* represented the slope of the cumulative lognormal collection efficiency curve associated with the $PM_{10}$ ambient air sampler. Based on Hinds (1982) definition, the primary performance characteristics for ambient $PM_{10}$ sampler used in the simulations will be a $d_{50}$ of 10 +/- 0.5 μm and a *slope* of 1.5 +/- 0.1. However, $d_{50}$ and *slope* values beyond these tolerances were used in estimating the inherent errors associated with sampler performance characteristics varying beyond established tolerances.

EPA's $PM_{2.5}$ ambient air sampler cumulative penetration data set produced a relatively smooth curve; however, the curve appeared to have a larger *slope* associated with particle sizes less than 2.5 μm AED than the *slope* associated with the particle sizes larger than 2.5 μm AED. It appears from the literature that EPA intended for the $PM_{2.5}$ sampler to have a "sharp cut" or represent a true concentration of $PM_{2.5}$ which would mean that, ideally, the *slope* would be equal to 1.0. However, from an engineering standpoint, it is not possible to design a sampler with a true cut. Work by Peters and Vanderpool (1996) suggested that the *slope* of 1.18 could be achieved with the WINS Impator, an EPA approved ambient air sampler. Further work by Buch (1999) suggested that the *slopes* were not as sharp as previously reported and that a more appropriate estimation of the sampler *slopes* would be 1.32 ± 0.03. Based on Buch's (1999) work, the primary performance characteristics for ambient $PM_{2.5}$ sampler used in the simulations will be a $d_{50}$ of 2.5 +/- 0.2 μm and a *slope* of 1.3 +/- 0.03. However, $d_{50}$ and *slope* values beyond these tolerances were used in estimating the inherent errors associated with sampler performance characteristics varying beyond established tolerances. Figure 12 graphically illustrates the differences between a $PM_{2.5}$ sampler-cut, $PM_{10}$ sampler-cut, TSP sampler-cut, $PM_{2.5}$ true-cut, and a $PM_{10}$ true-cut in relationship to a PSD characterized by a MMD of 20 μm and a GSD of 2.0.

The EPA $PM_{10}$ stack sampler will also be evaluated; however, the $PM_{2.5}$ stack sampler will not be evaluated because of the limited information available on the acceptable (EPA defined) sampler performance characteristics and tolerances. According to the USEPA (2002), the $PM_{10}$ stack sampler has a $d_{50}$ tolerance of 10.0 +/- 1.0 μm AED. The EPA does not explicitly state the *slope* tolerances associated with the $PM_{10}$ sampler; however, EPA does provide an efficiency envelope defining the acceptable ranges for the $PM_{10}$ stack sampler fractional collection efficiency curve, as shown in Figure 10. A $d_{50}$ range of 10.0 +/- 1.0 μm will be used in the simulations. A trial and error procedure will be used to determine the range of *slopes* that can be used with $d_{50}$ values of 9.0, 10.0 and 11.0 μm to produce a fractional collection efficiency curve that falls within the EPA define efficiency envelop. The *slopes* determined from the trial and error procedure will be used in the simulation.

The sampler performance characteristics previously defined for the $PM_{10}$ and $PM_{2.5}$ ambient air samplers and the $PM_{10}$ stack sampler were used in equation 16 to estimate the errors associated with the tolerances established for each of these samplers.

## Estimating Sampler and True Concentrations

Sampler concentrations can be theoretically estimated using PSD and sampler performance characteristics defined in equations 1 and 16, respectively, for particles described by a lognormal distribution. The method of determining sampler concentrations depends on whether the sampler uses a single or multi-stage pre-separator. For instance, most $PM_{10}$ ambient air samplers are single stage; however, an EPA approved $PM_{2.5}$ ambient air sampler consists of a $PM_{10}$ pre-separator and a $PM_{2.5}$ pre-separator. There are some $PM_{2.5}$ samplers that do not include the $PM_{10}$ pre-separator. Sampler concentrations for single stage samplers, $C_m(MMD,GSD,d_{50},slope)$, can be estimated by

$$C_m\left(MMD,GSD,d_{50},slope\right)=C_a\int_0^\infty f(d_p,MMD,GSD)P_m(d_p,d_{50},slope)dd_p \qquad (18)$$

Sampler concentrations for a two stage sampler, $C_{m2}(MMD,GSD,d_{501},slope_1,d_{502},slope_2)$, can be estimated by

$$C_{m_2}\left(MMD,GSD,d_{50_1},slope_1,d_{50_2},slope_2\right)=$$

$$C_a\int_0^\infty f(d_p,MMD,GSD)P_{m1}(d_p,d_{501},slope_1)P_{m2}(d_p,d_{502},slope_2)dd_p \qquad (19)$$

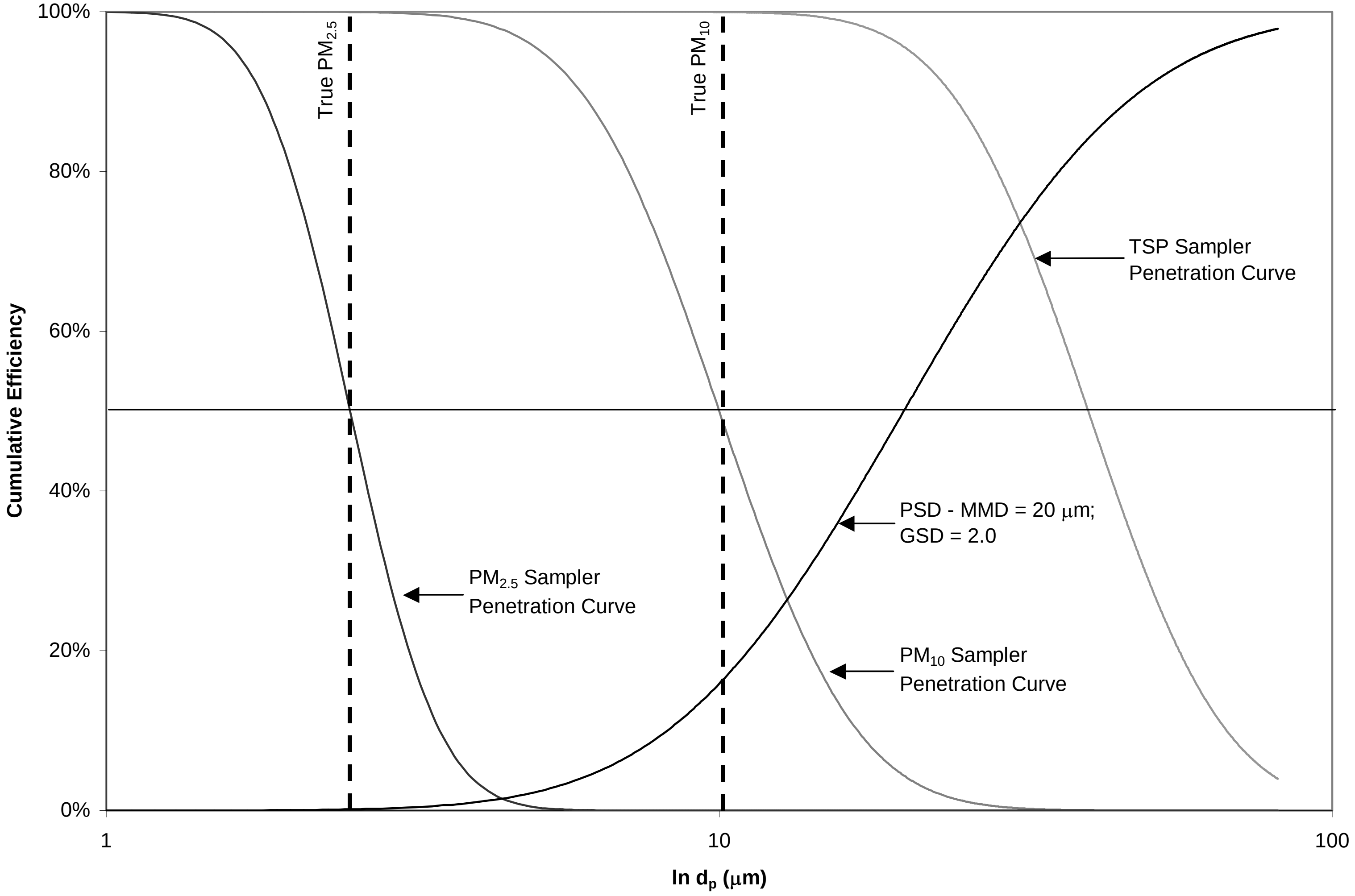


**Figure 12.** $PM_{2.5}$, $PM_{10}$, and TSP sampler penetration curves.

where $P_{m1}$ corresponds to the initial pre-separator and $P_{m2}$ corresponds to the secondary pre-separator.

For true concentrations, the cumulative penetration efficiency distribution function is assumed to be equal to 1 for all particle sizes less than or equal to the size of interest and zero for all other particle sizes. Therefore, the true concentration, $C_t(MMD,GSD,d_{50})$, can be estimated by

$$C_t\left(MMD,GSD,d_{50}\right) = C_a \int_0^{d_{50}} f(d_p,MMD,GSD)\,dd_p \tag{20}$$

If the PSD is described by a uniform distribution, equations 18, 19, and 20 will need to be further modified. For a uniformly distributed PSD, as described in equation 2, a single stage sampler concentration, $C_m(n,R,d_{50},slope)$, can be estimated by

$$C_m\left(n,R,d_{50},slope\right) = C_a \int_0^{\infty} f(d_p,n,R)P_m(d_p,d_{50},slope)\,dd_p \tag{21}$$

Likewise, a two stage sampler concentration, $C_{m2}(n,R,d_{501},slope_1,d_{502},slope_2)$, can be estimated by

$$C_{m_2}\left(n,R,d_{50_1},slope_1,d_{50_2},slope_2\right) =$$

$$C_a \int_0^{\infty} f(d_p,n,R)P_{m_1}(d_p,d_{50_1},slope_1)P_{m_2}(d_p,d_{50_2},slope_2)\,dd_p \tag{22}$$

where $P_{m1}$ corresponds to the initial pre-separator and $P_{m2}$ corresponds to the secondary pre-separator.

For true concentrations, $C_t(n,R,d_{50})$, equation 20, is modified as follows using equation 2:

$$C_t\left(n,R,d_{50}\right)=C_a\int_0^{d_{50}} f(d_p,n,R)dd_p \tag{23}$$

Relative Differences Between Sampler and True Concentrations

Sampler and true concentrations are not always equal. An estimate of the differences, *E(x)*, between these two concentrations is defined as

$$E(x)=\frac{(Measured-True)}{True}=\left(\frac{Measured}{True}\right)-1 \tag{24}$$

where *Measured* and *True* represent the estimated sampler and the true concentrations, respectively. Substituting equations 18 and 20 into equation 24 and canceling like terms, yields

$$E(MMD,GSD,d_{50},slope)+1=\left[\frac{\int_0^{\infty} f(d_p,MMD,GSD)P_m(d_p,d_{50},slope)dd_p}{\int_0^{d_{50}} f(d_p,MMD,GSD)dd_p}\right] \tag{25}$$

for a sampler with a single pre-separator. Equation 25 can further expanded for a multi-stage pre-separator sample in the same manner in which equation 18 was expanded. *E(MMD, GSD,* $d_{50}$*, slope)+1* will be referred to as the ratio of the sampler to true concentration. Equation 25 and the corresponding equation for a multi-stage pre-separator sampler were solved for various PSD and sampler performance characteristics in order estimate the errors associated with the interaction of these two characteristics.

### Cotton Gin Exhaust PSD Estimates

The best method for determining the PSD characteristics associated with the various cotton gin exhausts is to conduct stack sampling on each individual exhaust of several cotton gins. A cotton gin material handling system flow diagram is shown in Figure 13. However, because of the cost and other considerations, an alternative method was selected. This method included determining the PSD characteristics associated with PM less than 100 μm contained in cotton gin trash processed by various process streams, determining the PSD characteristics associated filters collected during two commercial stack sampling tests (limited exhausts tested), and estimating PSD characteristics based on EPA's 1996 AP-42 list of cotton gin emission factors. A Coulter Counter Multisizer III was used in performing all PSD analysis.

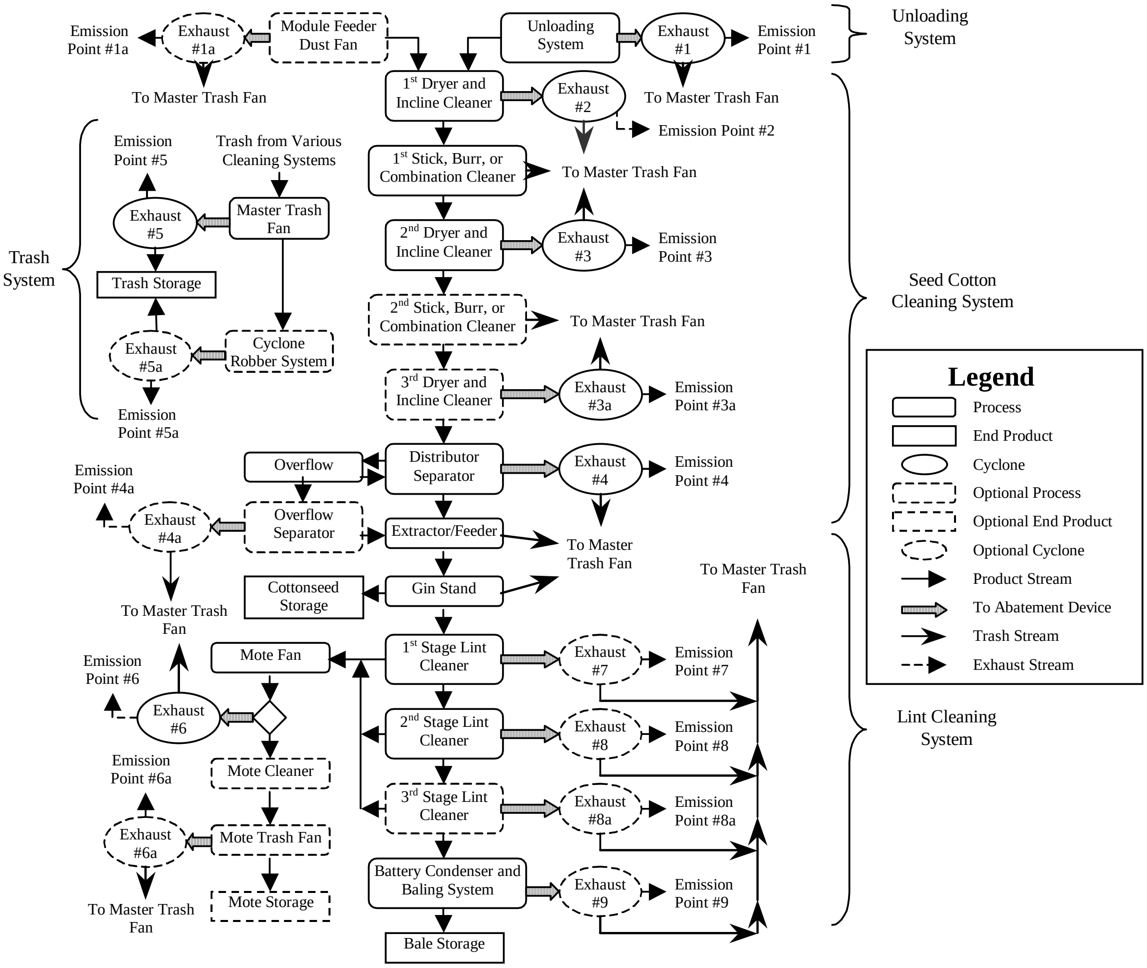


**Figure 13.** Cotton gin material handling system flow diagram.

## Coulter Counter Analysis

In the 1940's, William Coulter developed and patented a technique that allowed particles homogenously suspended in a conducting liquid to be simultaneously counted and sized. It was originally developed for the use in hospitals for performing blood cell counts but is being increasingly used in other technical applications (Richards, 1968). This technique is known as the Coulter Principle, or the electrical sensing zone method. With this method, PM is dispersed in an electrically conductive fluid (electrolyte). This electrolyte is forced through a small aperture in an insulated wall with a high precision metering pump (Beckman Coulter, 2000). Electrodes located on either side of the aperture produces a constant, controlled electric current flow through the aperture. As each particle suspended in the electrolyte enters the aperture it displaces a volume of electrolyte equal to its own volume. This momentarily increases the impedance across the aperture tube. The increased impedance produces a current flow into an amplifier. The current fluctuation is converted into a voltage pulse that is directly proportional to the volume of the particle. The pulses generated by the particles are counted and the pulse height is analyzed to determine particle volume. The pulse data can be stored in up to 300 channels (user-defined). A size spectrum can be acquired by scaling these pulse heights in measured units. The Coulter process is illustrated in Figure 14. A Beckman Coulter Counter Multisizer III was used for all PSD analysis.

The Multisizer III provides both particle counting and sizing within an overall size range of 0.4 to 1200 μm, dependent on aperture tube size. A 100 μm aperture tube was used in all analysis which corresponds to a particle size range of 2 to 60 μm equivalent spherical diameter (ESD). The Multisizer III provides the option for the PSD to be determined based on elapsed time, precise volumes, or particle count. For this research, a particle count 300,000 was used for all analysis. Results from the Multisizer III particle size analysis were PSDs in the form of particulate volume or number versus ESD.

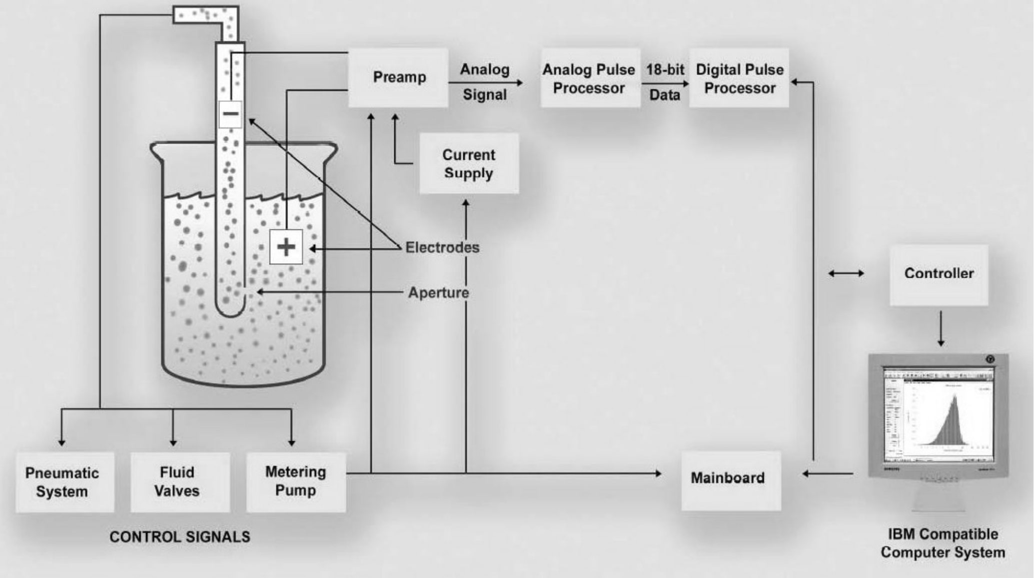


**Figure 14.** Illustration of the Coulter process (Beckman Coulter, 2000).

The electrolyte used in the Coulter Counter analysis was a 5% lithium chloride/methanol solution. The electrolyte was pre-filtered using a filtration system that removed all particles larger than 0.2 μm. A background count of the filtered electrolyte was made with the Coulter Counter to ensure minimal particulate contamination of the electrolyte. A background count of less than 200 particles per three $cm^3$ was viewed as acceptable.

The following procedures were used in determining the PSD of PM captured on filters used in this study:

1) Two 3.8 $cm^2$ round cutouts were collected from a filter (used for poly-web filter media only). The cutouts were placed in a beaker containing approximately 40 ml of pre-filtered electrolyte. The PM was dispersed in the electrolyte by exposing the solution to an ultrasonic bath for fifteen minutes. When analyzing PM collected on glass fiber filters, the same procedure was used except the PM sample from the filter was collected by rolling a nylon swab across the filter. A

more detail explanation of why two different procedures were used on the poly-web and glass fiber filter media is contained in Appendix B.

2) The electrolyte containing the dispersed PM was passed through a nylon 100 μm monofilament screen.
3) A sub-sample of the dispersed solution is slowly added to pre-filtered electrolyte contained in the Multisizer III beaker. The final concentration of PM in the Multisizer beaker should be between 6 and 10%. The concentration is limited in order to keep occurrence of coincidence low (more than one particle being counted as a single particle).
4) Once the PSD analysis is completed, the Multisizer beaker is cleaned and loaded with filtered electrolyte.
5) Steps 3 and 4 are replicated three times for the solution prepared in steps 1 and 2.

As previously stated, Coulter PSDs are based on ESD. In order to convert the Coulter based PSD to PM mass percent versus AED, the following equations can be used:

$$d_a^2\left[1+\frac{\lambda}{d_a}\left(2.514+0.8e^{\left(-0.55\frac{d_a}{\lambda}\right)}\right)\right]=\frac{\rho_p d_p^2}{\kappa\rho_w}\left[1+\frac{\lambda}{d_p}\left(2.514+0.8e^{\left(-0.55\frac{d_p}{\lambda}\right)}\right)\right] \tag{26}$$

where $d_a$ = aerodynamic equivalent diameter,
$d_p$ = particle diameter (ESD),
$\lambda$ = mean free path, 0.066,
$\rho_p$ = particle density, and
$\rho_w$ = density of water,
$\kappa$ = dynamic shape factor.

Equation 26 can be simplified for particle diameters larger than about 2.0 μm to

$$d_a = d_p\left(\frac{\rho_p}{\kappa\rho_w}\right)^{1/2} \tag{27}$$

$\rho_w$ is the density of water with a value of 1 g/cm$^3$. $\rho_p$ is the particle density, and was assumed to be constant for the various size particles with in a given sample. A pycnometer is a quick and accurate method of determining average particle density. The Micromeritics AccuPyc 1330 has a reported accuracy of 0.03% of reading plus 0.03% of sample capacity, and was used in this research. The AccuPyc 1330 pycnometer uses a precision-calibrated volume chamber and uses helium as the displacement medium. PM is placed in the AccuPyc chamber, the chamber is sealed, helium is inserted into the chamber, and the PM sample displaces some of the helium molecules thereby changing the pressure in the sample chamber. The change in pressure is a direct measure of the PM volume. This measure of PM volume is coupled with the mass of the material placed in the chamber in order to determine the average particle density.

Particulate materials from natural and manmade sources are often nonspherical in shape. The drag force on a nonspherical particle is generally greater than that on a sphere of the same volume moving at the same velocity (Cheng et al., 1988). Therefore, the behavior of a particle is determined by particle size, shape, and density. The dynamic shape factor, $\kappa$, relates the sedimentation diameter to the equivalent volume diameter and is defined as

$$\kappa = \frac{d_v^2}{d_s^2} \tag{28}$$

where $d_s$ is the sedimentation diameter and $d_v$ is the volume diameter. Dynamic shape factors generally range from 1.0 to 2.0, with spherical particles have a dynamic shape factor of 1.0. There are currently no dynamic shape factor estimates for cotton gin PM. Particles associated with cotton gin exhausts were evaluated under a scanning electron microscope. An example photograph is shown in Figure 15.

Based on the apparent particle shape variability associated with cotton gin exhaust PM, an assumption was made to set the dynamic shape factor equal to 1.0 for all samples. By reporting all results with a dynamic shape factor of 1.0, the data produced in this research could be easily modified to incorporate various dynamic shape factors at a later time. According to Hinds (1999), the dynamic shape factors for quartz and sand dusts are 1.36 and 1.57, respectively. If these dynamic shape factors were assumed for the cotton gin PM, then reporting the results of this research with a dynamic shape factor of 1.0 would result in the particle sizes being over-estimated by 17% and 25% if the dynamic shape factors were similar to quartz and sand, respectively. Therefore, based on the lack of a dynamic shape factor estimates for cotton gin PM, the apparent particle shape variability associated with the cotton gin PM, and the potential over-estimations, it was determined that incorporating a dynamic shape factor of 1.0 was the most appropriate method of reporting the results.

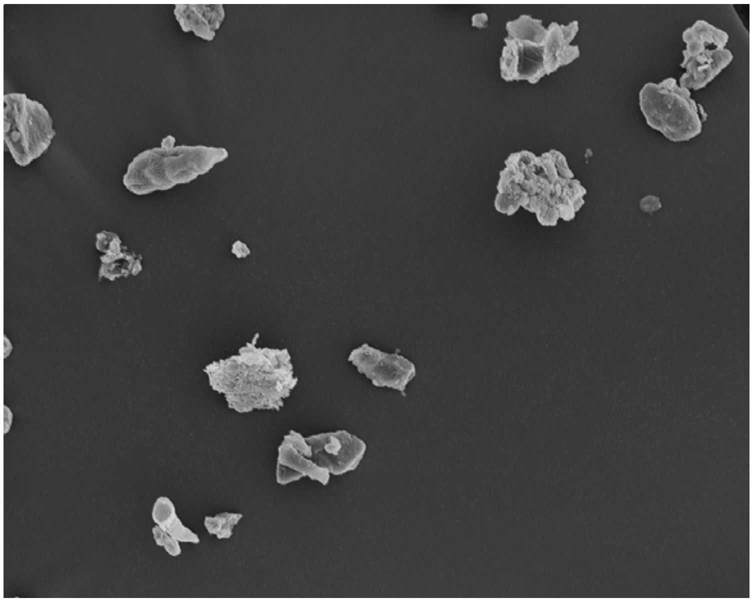


**Figure 15.** Scanning electron microscope photograph of cotton gin exhaust particles.

### Cotton Gin Trash PSDs for Various Process Streams

Cotton gin trash samples from various machines and process streams were obtained from the USDA-ARS Cotton Ginning Research Unit in Stoneville, MS, and the USDA-ARS Cotton Production and Processing Research Unit in Lubbock, TX. The samples received from the Stoneville Laboratory corresponded to picker and stripper cotton varieties (i.e. the seed cotton was from mixed varieties that were either picker or stripper harvested). The seed cotton was ginned in the Stoneville Laboratories micro-gin. A unique feature of this gin is that all the material removed from the individual machines is dropped out below the machine into a catch pan. Generally, this material would be picked up by an air stream and processed by a cyclone or some other abatement device. The samples received from the Stoneville Laboratory corresponded to the 1st and 2nd incline cleaners, 1st stick machine, 2nd stick machine for the stripper cotton, gin stand and mote system for the picker cotton, and lint cleaners. Each of the samples was air washed and the material removed from each sample was collected on an 8” x 10” poly-web filter. The air wash device is essentially composed of a sample chamber that has an outside skin composed of a 100 μm mesh screen. Air is pulled through a pipe running through the center of the chamber, through the sample, and through the poly-web filter by a fan. The camber is continuously rotated for 15 minutes, while the fan is pulling the air through the sample. The particle size distribution and particle density of the material captured on the filter were then determined using the previously defined procedures.

The samples received from the USDA-ARS Cotton Processing and Production Research Unit were samples collected during a study conducted by Holt et al. (2000). Results of this study were highlighted in Tables 7-9. The samples received from the Lubbock laboratory corresponded to Paymaster HS 26 and HS 200 cotton varieties. Both varieties were stripper harvested with and without field cleaning. Each of the samples corresponded to one bale lots, and three replications were received from each treatment. Although Holt et al. (2000) collected material from each process stream; samples from similar process streams were combined. Therefore, the samples received corresponded to the unloading system, feeder and gin stand, incline cleaners, extractors, and lint cleaners. Each of the samples was air washed and the material removed from each sample was collected on an 8” x 10” poly-web filter. The particle size distribution and particle density of the material captured on the filter were then determined using the previously defined procedures.

The material received from both the Lubbock and Stoneville laboratories consisted of three replicated samples for each treatment. Three sub-samples were collected from each sample received. Particle size analyses and particle densities were determined for each sub-sample. In addition, particle size analyses consisted of three runs per sub-sample. The Proc Mixed (Littell et al., 1996) procedure in SAS was utilized to compare the PSD characteristics and particle densities associated with the samples received from each laboratory, individually.

### Commercial Stack Sampling

The USDA-ARS Southwestern Cotton Ginning Research Laboratory in Mesilla Park, NM, was involved in two cotton gin stack sampling tests conducted in 2001. The stack sampling tests were conducted at the Idria Gin #1 (roller gin) in Five Points, CA, and the Mesa Farmers Cooperative Gin in Mesquite, NM. Sampling was conducted by two separate independent sampling companies. The New Mexico ginning laboratory arranged for the filters and wash from each of the sampling tests to be used in this research.

The primary objective of the Idria stack sampling test was to determine if there were differences in PM emission factors for first and second pick Pima cotton. The gin’s #1 pre-cleaning exhaust was selected for source sampling. This system’s exhaust is controlled by a set of four 1.12 m (44 inch) diameter 1D-3D cyclones, equipped the traditional 1D-3D inlets. This is the gin’s second exhaust, and handles heated air from the first seed cotton drier. The #1 pre-cleaning system incorporates an incline cleaner and a stick machine to remove waste from the seed cotton. Typically, this is one of the heavier loaded high-pressure gin exhausts in terms of PM emissions.

A candy cane shaped circular duct was attached to the exit tube of one of the four 1D-3D cyclones and continued vertically to the ground. Source sampling was conducted through two sample ports cut in the candy cane duct. The sample ports were located three diameters upstream and eight diameters downstream from the nearest air flow disturbance. A 12-point traverse sampling scheme (six points per port) was utilized for each sampling run in accordance with EPA's Mehtod 201a guidelines.

An EPA Method 201a sampling train was utilized to determine the mass of TSP and $PM_{10}$ emitted from the cyclone selected for testing. The sample train consisted of a stainless steel nozzle, stainless steel Anderson $PM_{10}$ cyclone separator, glass fiber filter, stainless steel probe, and cooled impingers. The sampling train can be divided into three essential components in regards to PM mass: cyclone wash, post-cyclone wash, and filter. In order to determine the TSP mass, all three mass components were added together. $PM_{10}$ mass was determined by adding the mass of the post-cyclone wash and the mass of PM on the filter. All tests were conducted isokinetically. Six test runs were performed for both the first and second pick Pima cotton, for a total of twelve test runs.

Idria Gin #1's production data were used to determine bale production rates in terms of the number of 227 kg (500 lb) bales per hour. Production rates were kept as constant as possible during the tests, with source testing conducted over a two-day period. This timing was used so that, with stable weather conditions, both cottons would be processed under the same environmental conditions. All TSP and $PM_{10}$ emission factors for the source sampling tests were calculated using the following equation:

$$EF = \frac{mass_{PM} * Area_{Exhaust} * CF}{Rate_{Processing} * time_{sampling} * Area_{Nozzle} * IsokineticVariation} \tag{29}$$

where,

$EF$ = emission factor, kg/bale (lb/bale);

$mass_{PM}$ = mass of PM, g;

$Area_{Exhaust}$ = exhaust area, $m^2$ ($ft^2$);

$Area_{Nozzle}$ = nozzle area, $cm^2$ ($ft^2$);

$Rate_{Processing}$ = processing rate, bales/h;
$Time_{sampling}$ = sampling time, min; and
$CF$ = conversion factor, 600 (0.132).

TSP and $PM_{10}$ emission factors for the first and second pick Pima cotton were determined for each test run. The emission factors for each run were multiplied by four (number of cyclones associated with the system) to obtain a total emission factor for the #1 pre-cleaning system. $AIR_x$ Testing, Ventura, CA, certified source-test contractors, conducted the source sampling and data reduction.

Particle size analysis was conducted on each of the filters and washes received from $AIR_x$ Testing using the procedures discussed in a previous section. The percent of particles less than 10 and 2.5 μm, determined from the particle size analyses for the cyclone wash, post-cyclone wash, and filters, were multiplied by the corresponding masses of PM reported by $AIR_x$ Testing to determine the true $PM_{10}$ and $PM_{2.5}$ masses. The mass corresponding to $PM_{10}$ for the cyclone wash, post-cyclone, and filters were added together in order to determine the true total $PM_{10}$ mass. The same process was used to determine the true total $PM_{2.5}$ mass. Equation 29 and total $PM_{10}$ and $PM_{2.5}$ masses were used to determine the corresponding emission factors. These emission factors were multiplied by four to determine the true total emission factors for Idria Gin #1's first pre-cleaning system.

The State of New Mexico, Environmental Department, Air Quality Bureau required that stack sampling tests be conducted at the Mesa Farmers Cooperative Gin for permitting purposes. The gin's unloading, 1st pre-cleaning, 2nd pre-cleaning, 3rd incline (systems A & B), and the lint basket pull systems were selected for source sampling. These system exhausts are controlled by multiple 1D-3D cyclones with inverted inlets. All exhausts utilized two 1D-3D cyclones, except for the lint basket pull system that only used one cyclone. The cyclone diameters for the unloading, 1st pre-cleaning, 2nd pre-cleaning, 3rd incline (systems A & B), and the lint basket pull systems were 1.37 m (56 in.), 1.22 m (48 in.), 1.22 m (48 in.), 0.81 m (32 in.), 0.81 m (32 in.), and 0.97 m (38 in.), respectively. Method 5 (TSP) and Method 201a ($PM_{10}$) sampling was conducted one cyclone from each exhaust.

The sampling procedures used were similar to those used for the Idria Gin #1 testing, with the exception of including the Method 5 sampling. The Method 5 sampling protocol used was similar to the Method 201a sampling protocol, except the $PM_{10}$ sampling cyclone is excluded from the sampling train. For Method 5 sampling, the sampling train can be divided into two essential components in regards to PM mass: wash and filter. In order to determine the TSP mass, both mass components are added together. Three test runs were performed for all exhausts, for a total of 18 Method 5 and 18 Method 201a test runs.

Production data and calculation of emission factors were completed in the same manner as that conducted for the Idria Gin #1. Energy & Environmental Measurement Corporation (EEMC) in Tucson, AZ, certified source-test contractors, conducted the source sampling and data reduction. Particle size analysis and determination of true $PM_{10}$ and $PM_{2.5}$ emission factors were completed using the same procedures used for the Idria Gin #1, except that the individual exhaust emission factors were multiplied by the corresponding number of cyclones associated with the exhaust.

Results from the Idria test consisted of six replicated samples for each treatment and results from the Mesa test consisted of three replicated samples for each treatment. The particle size analyses consisted of three runs per sample. The Proc Mixed (Littell et al., 1996) procedure in SAS was utilized to compare the PSD characteristics and emission factors associated with the samples received from each test.

### PSDs Estimated from AP-42 Emission Factors

The number and type of process streams associated with cotton gin systems will vary from gin to gin. A process steam refers to a sequence of one or more process that is followed by an exhaust. A cotton gin material handling system flow diagram was provided in Figure 13, which includes the basic process streams found in virtually all gins and optional streams that may or may not be associated with a particular cotton gin. The basic streams include: 1) unloading, either suction or module feeder; 2) 1st stage of seed cotton cleaning; 3) 2nd stage of seed cotton cleaning; 4) distributor/overflow; 5)1st stage of lint cleaning; 6) 2nd stage of lint cleaning; 7) battery condenser; 8) mote; and 9) trash. Optional process streams that may be incorporated in a particular cotton gin are: 1) 3rd stage of seed cotton cleaning; 2) overflow separator; 3) 3rd stage of lint cleaning; 4) mote cleaning; and 5) cyclone robber.

The EPA published emission factors for cotton gins in AP-42 (USEPA, 1996b) that are commonly used as guidelines in the permitting process if actual source sampling data is not available for a particular cotton gin. The 1996 AP-42 TSP and $PM_{10}$ emission factors are listed in Table 11. Within the 1996 AP-42, EPA provides emission factors for virtually all cotton gin exhausts illustrated in Figure 13. Process stream exhaust emission factors not incorporated (shown as NR in Table 11) in the 1996 AP-42 document are: 1) TSP and $PM_{10}$ values for the module feeder and overflow separator; 2) $PM_{10}$ values for lint cleaners and battery condensers with covered condenser drums; and 3) TSP and $PM_{10}$ values for individual lint cleaner exhausts (i.e. TSP and $PM_{10}$ emission factors are combined for 1st and 2nd stage lint cleaners).

The TSP and $PM_{10}$ emission factors listed in the 1996 AP-42 were used to systematically solve equations 18 and 20 through a trial and error process. Mathcad 2002 was used to carryout the mathematical procedure. In order to solve the equations the following broad assumptions were made:

1) emission factors presented in Table 11 represent typical values that can be expected from an average cotton gin;
2) the AP-42 emission factors are based on Method 201a stack sampling data;
3) the Method 201a $PM_{10}$ sampling cyclone performance characteristics exhibited during the collection of the AP-42 data were within EPA's defined tolerances (i.e., a $d_{50}$ of 10 +/- 1.0 μm and a fractional collection efficiency curve that falls within EPA's defined collection efficiency envelope, shown in Figure 9); and
4) the PSD of the dust exiting the exhaust abatement devices can be described by a lognormal distribution.

These assumptions provide the mathematical basis for using equations 18 and 20 to calculate sampler and true concentrations.

There are four unknowns in equation 18; therefore, additional assumptions were required. Based on the EPA $PM_{10}$ sampling cyclone $d_{50}$ criteria, a $d_{50}$ value of 11.0 μm was assumed. This $d_{50}$ value corresponds to the upper limit defined by EPA. The *slope* associated with the $PM_{10}$ sampling cyclone will be determined from results stemming from objective one of this research. In order to further simplify this procedure, three *GSD* values will be assumed based on the PSD results from the gin trash evaluation and stack sampling evaluation conducted in objective two of this research. Based on these assumptions, equation 18 will be solved through a trial and error process until the calculated sampler concentration equals the 1996 AP-42 $PM_{10}$ to TSP emission factor ratio. This process was completed for all exhausts listed in the 1996 AP-42.

The *MMD* and *GSD* values obtained from the trial and error procedure, using equation 18, will be used to determine the corresponding true $PM_{10}$ emission factors based on individual process stream exhaust PSD characteristics. Equation 20, with a $d_{50}$ value of 10.0 μm, was used to calculate the true $PM_{10}$ percentages associated with the resulting PSD characteristics. The $PM_{10}$ percentages were then multiplied by the corresponding 1996 AP-42 TSP emission factor in order to calculate the true $PM_{10}$ emission factor for each exhaust. This process was completed for all exhausts listed in the 1996 AP-42.

In addition to determining the PSD characteristics associated with the cotton gin exhausts listed in the 1996 AP-42, a weighted average PSD will be generated for each of the assumed *GSDs*. The average PSD characteristics will be determined by adding all the PSDs associated with the process stream exhausts and characterizing the resulting PSD as a lognormal distribution. In this process, the 1996 AP-42 individual process stream emission factors are multiplied by the process stream mass density function and then that value is divided by the total 1996 AP-42 TSP emission factor. This process will be completed for particle diameters ranging from 0 to 200 μm in increments of 0.01 μm. This series of values will be compared to a lognormally distributed PSD described by the average *MMD* and *GSD*. The mass density function associated with the average PSD will cover particle diameters ranging from 0 to 200 μm, in increments of 0.01 μm. The absolute difference between the summed process stream values and the average values will be calculated for each individual particle size bin. This difference will be summed. A trial and error process will be completed to minimize (close to zero) the summed difference in order to estimate the average *MMD* and *GSD*. Equation 20 will then be used to determine the true $PM_{10}$ percentage associated with the average PSD. This true average $PM_{10}$ percentage will then be multiplied by the 1996 AP-42 TSP emission factors in order to determine the true $PM_{10}$ emission factors based on an average PSD.

# RESULTS AND DISCUSSION

The results of this research are broken down by objective and further segregated by secondary topics. The main sections, corresponding to the objectives, are inherent sampler errors and cotton gin exhaust PSD estimates.

## Inherent Sampler Errors

The inherent sampler error findings and corresponding discussions are broken down into several sub-sections. The sections include: 1) errors attributed to established sampler performance tolerances, for the ambient $PM_{10}$ and $PM_{2.5}$ samplers and the $PM_{10}$ stack sampler; 2) errors attributed to the interaction of particle size and sampler performance characteristics (discussed in terms of ambient $PM_{10}$ samplers, ambient $PM_{2.5}$ samplers, ambient $PM_{coarse}$, and the ratio of ambient $PM_{2.5}$ to ambient $PM_{10}$); and 3) errors attributed to sampler performance characteristics varying beyond the established tolerances (discussed in terms of ambient $PM_{10}$ samplers, ambient $PM_{2.5}$ samplers, ambient $PM_{coarse}$, and the ratio of ambient $PM_{2.5}$ to ambient $PM_{10}$).

### Ambient $PM_{10}$ Sampler Performance Characteristics

EPA essentially defines the $d_{50}$ and slope associated with the $PM_{10}$ ambient air sampler in 40 CRF, Part 53 in the discussion of tests required for a candidate sampler to receive EPA approval. The $d_{50}$ for the $PM_{10}$ sampler is explicitly stated in the EPA standards as 10.0 +/- 0.5 μm AED. No slope values for the sampler are listed in 40 CFR, Part 53 or any other current EPA standard; however, penetration data is presented in 40 CFR Part 53. Ideally, the penetration data could be fit to a cumulative lognormal distribution to determine the characteristic $d_{50}$ and slope for the $PM_{10}$ samplers; however, it was found that no single cumulative lognormal curve adequately represented the EPA data set in 40 CFR, Part 53. It should be noted that this penetration data, along with EPA defined interval mass concentrations and mass penetration tolerances, are used to determine if proposed samplers meet EPA's $PM_{10}$ performance criteria.

According to the literature, the ideal ambient $PM_{10}$ sampler penetration curve should mimic the thoracic fraction of PM deposited in the human respiratory system. In Figure 16, the EPA's ideal $PM_{10}$ ambient sampler penetration data is overlaid on the ACGIH sampling criteria for the thoracic fraction of PM. Based on the curves in Figure 16, the ambient $PM_{10}$ sampler penetration data appears to follow the thoracic convention fairly well for particle sizes less than about 13 μm AED. For particle diameters larger than 13 μm AED, the cumulative collection efficiency for EPA's ambient $PM_{10}$ sampler penetration data moves towards zero much more rapidly than the thoracic penetration convention.

The $PM_{10}$ cumulative penetration data set produced a rough curve which appeared to have a larger slope for the particle sizes less than 10 μm than the slope for the particle sizes greater than 10 μm. Hinds (1982) suggested that the slope associated with PM deposited in the human respiratory system had a slope of 1.5 +/- 0.1 and that this slope represented the slope of the cumulative lognormal collection efficiency curve associated with the $PM_{10}$ ambient air sampler. For the purposes of this research, the $PM_{10}$ sampler performance characteristics were defined as $d_{50}$ of 10 +/- 0.5 μm and a slope of 1.5 +/- 0.1.

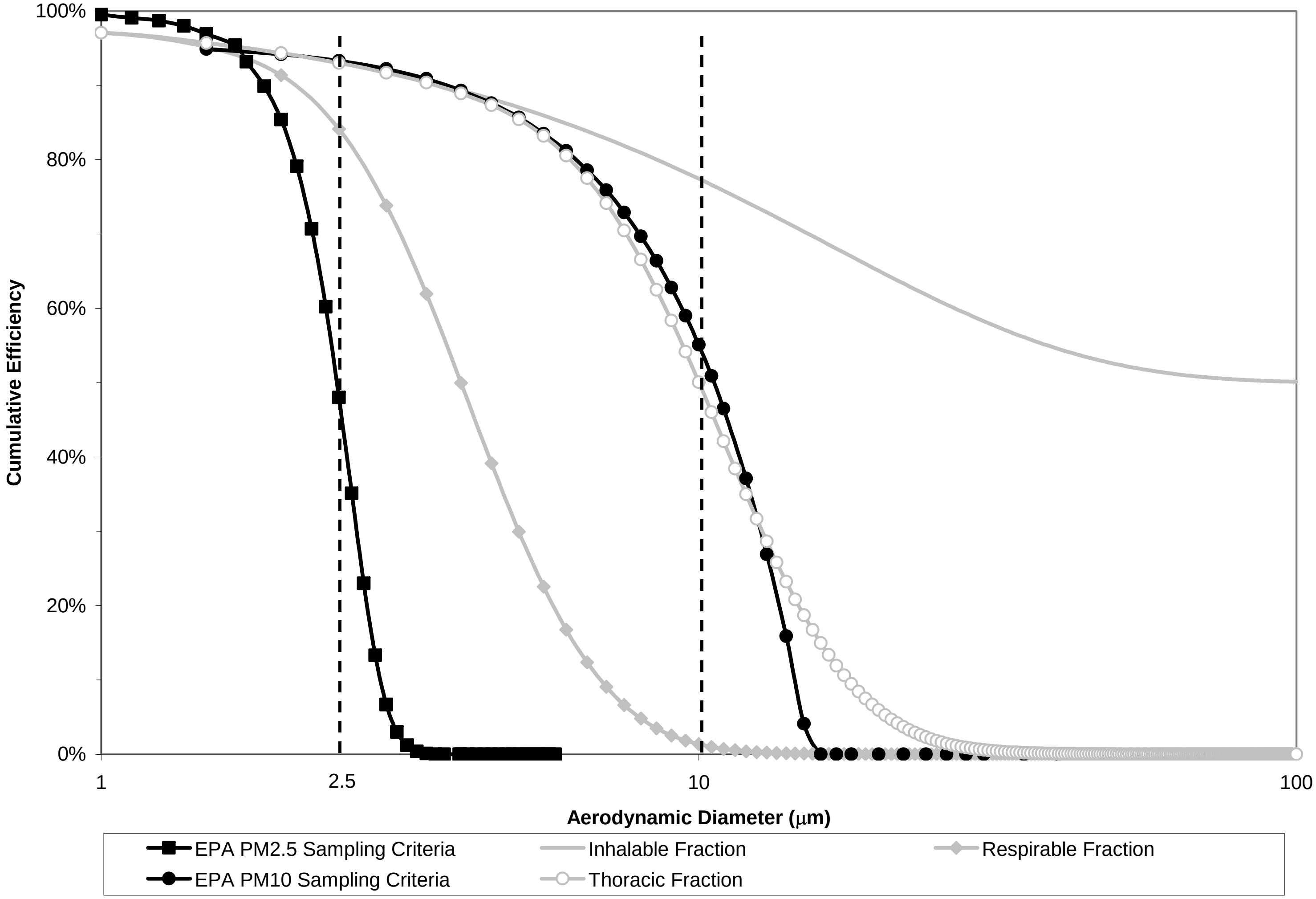


**Figure 16.** The EPA ideal $PM_{10}$ and $PM_{2.5}$ sampler penetration curves overlaid on the ACGIH sampling criteria for inhalable, thoracic and respirable fractions of PM (ACGIH, 1997; *CFR*, 2001e).

The performance characteristic ranges used to define the ambient $PM_{10}$ sampler performance characteristics in this research were divided into nine $d_{50}$ and slope combinations: all combinations for $d_{50}$ values of 9.5, 10.0, and 10.5 μm and slope values of 1.4, 1.5, and 1.6. These nine sampler performance criteria were evaluated using the EPA wind tunnel evaluation guidelines for the ambient $PM_{10}$ sampler. The procedure included: 1) the determination of penetration efficiency (referred to as "sampling effectiveness" by the EPA) for a specific set of sampler performance criteria for the particle sized defined by the EPA; 2) the penetration efficiency for each particle size was multiplied by the interval mass concentration defined by the EPA in order to determine an expected mass concentration; and 3) the expected mass concentration was summed for all particle sizes and compared to the ideal sampler expected mass concentration defined by the EPA. The calculation values used in determining the expected mass concentration for a $PM_{10}$ sampler with a $d_{50}$ of 10 μm and a slope of 1.5 are shown in Table 13. According to 40 CFR, Part 53, a candidate sampler passes the sampling effectiveness test if the expected mass concentration calculated for the candidate sampler differs by no more than +/- 10% from that predicted for the ideal sampler (*CFR*, 2001e). The results of the comparison on the nine sampler performance criteria used in this research to that of EPA's ideal sampler are shown in Table 14. Based on EPA's criteria of acceptance, all nine sampler performance criteria used in this research are acceptable (i.e. the sampling effectiveness for all nine sampler performance criteria were within +/- 10% of EPA's ideal sampler).

Based on the ambient $PM_{10}$ sampler performance criteria used in this research, four combinations of $d_{50}$ and slope values were used to define boundary penetration efficiency curves. These penetration curves were defined with $d_{50}$ values of 9.5 and 10.5 μm and slope values of 1.4 and 1.6. Figure 17 illustrates the comparison of the boundary penetration curves used in this research and EPA's ideal $PM_{10}$ sampler penetration efficiency curve. The ideal penetration curve is encompassed by the boundary penetration curves for particle diameters ranging from 6.5 to 14.5 μm.

**Table 13.** Expected mass concentration for a PM$_{10}$ sampler with a cutpoint of 10 µm and a slope of 1.5 and the EPA ideal PM$_{10}$ sampler in accordance with the EPA wind tunnel evaluation guidelines (*CFR*, 2001e).

| | Test Sampler | | | Ideal Sampler | | |
|---|---|---|---|---|---|---|
| Particle Size (µm) | Sampling Effectiveness | Interval Mass Concentration (µg/m$^3$) | Expected Mass Concentration (µg/m$^3$) | Sampling Effectiveness | Interval Mass Concentration (µg/m$^3$) | Expected Mass Concentration (µg/m$^3$) |
| < 1.0 | 1.000 | 62.813 | 62.813 | 1.000 | 62.813 | 62.813 |
| 1.5 | 1.000 | 9.554 | 9.554 | 0.949 | 9.554 | 9.067 |
| 2.0 | 1.000 | 2.164 | 2.164 | 0.942 | 2.164 | 2.038 |
| 2.5 | 1.000 | 1.785 | 1.784 | 0.933 | 1.785 | 1.665 |
| 3.0 | 0.999 | 2.084 | 2.081 | 0.922 | 2.084 | 1.921 |
| 3.5 | 0.995 | 2.618 | 2.605 | 0.909 | 2.618 | 2.380 |
| 4.0 | 0.988 | 3.211 | 3.173 | 0.893 | 3.211 | 2.867 |
| 4.5 | 0.976 | 3.784 | 3.691 | 0.876 | 3.784 | 3.315 |
| 5.0 | 0.956 | 4.300 | 4.112 | 0.857 | 4.300 | 3.685 |
| 5.5 | 0.930 | 4.742 | 4.409 | 0.835 | 4.742 | 3.960 |
| 6.0 | 0.896 | 5.105 | 4.575 | 0.812 | 5.105 | 4.145 |
| 6.5 | 0.856 | 5.389 | 4.613 | 0.786 | 5.389 | 4.236 |
| 7.0 | 0.810 | 5.601 | 4.540 | 0.759 | 5.601 | 4.251 |
| 7.5 | 0.761 | 5.746 | 4.373 | 0.729 | 5.746 | 4.189 |
| 8.0 | 0.709 | 5.834 | 4.136 | 0.697 | 5.834 | 4.066 |
| 8.5 | 0.656 | 5.871 | 3.850 | 0.664 | 5.871 | 3.898 |
| 9.0 | 0.603 | 5.864 | 3.533 | 0.628 | 5.864 | 3.683 |
| 9.5 | 0.550 | 5.822 | 3.204 | 0.590 | 5.822 | 3.435 |
| 10.0 | 0.500 | 5.750 | 2.875 | 0.551 | 5.750 | 3.168 |
| 10.5 | 0.452 | 5.653 | 2.556 | 0.509 | 5.653 | 2.877 |
| 11.0 | 0.407 | 8.257 | 3.361 | 0.465 | 8.257 | 3.840 |
| 12.0 | 0.326 | 10.521 | 3.435 | 0.371 | 10.521 | 3.903 |
| 13.0 | 0.259 | 9.902 | 2.563 | 0.269 | 9.902 | 2.664 |
| 14.0 | 0.203 | 9.250 | 1.881 | 0.159 | 9.250 | 1.471 |
| 15.0 | 0.159 | 8.593 | 1.363 | 0.041 | 8.593 | 0.352 |
| 16.0 | 0.123 | 7.948 | 0.979 | 0.000 | 7.948 | 0.000 |
| 17.0 | 0.095 | 7.329 | 0.699 | 0.000 | 7.329 | 0.000 |
| 18.0 | 0.074 | 9.904 | 0.729 | 0.000 | 9.904 | 0.000 |
| 20.0 | 0.044 | 11.366 | 0.496 | 0.000 | 11.366 | 0.000 |
| 22.0 | 0.026 | 9.540 | 0.247 | 0.000 | 9.540 | 0.000 |
| 24.0 | 0.015 | 7.997 | 0.123 | 0.000 | 7.997 | 0.000 |
| 26.0 | 0.009 | 6.704 | 0.062 | 0.000 | 6.704 | 0.000 |
| 28.0 | 0.006 | 5.627 | 0.031 | 0.000 | 5.627 | 0.000 |
| 30.0 | 0.003 | 7.785 | 0.026 | 0.000 | 7.785 | 0.000 |
| 35.0 | 0.001 | 7.800 | 0.008 | 0.000 | 7.800 | 0.000 |
| 40.0 | 0.000 | 5.192 | 0.002 | 0.000 | 5.192 | 0.000 |
| 45.0 | 0.000 | 4.959 | 0.001 | 0.000 | 4.959 | 0.000 |
| | | $C_{sam(exp)}$) | 150.646 | | $C_{ideal(exp)}$ | 143.890 |

**Table 14.** Estimated $PM_{10}$ mass concentration ratio between sampler performance characteristics and the EPA idealized sampler.

| $d_{50}$ (μm) | Slope | Ratio (%) |
|---|---|---|
| 9.5 | 1.4 | 100 |
| 9.5 | 1.5 | 101 |
| 9.5 | 1.6 | 102 |
| 10.0 | 1.4 | 104 |
| 10.0 | 1.5 | 105 |
| 10.0 | 1.6 | 106 |
| 10.5 | 1.4 | 107 |
| 10.5 | 1.5 | 108 |
| 10.5 | 1.6 | 109 |

When comparing the boundary penetration efficiency curves in Figure 18, it is apparent that there is an acceptable range of penetration efficiencies for the $PM_{10}$ ambient air sampler. The acceptable range of penetration efficiencies for a particle size of 10 μm AED is 44 to 56%, whereas the acceptable range for a particle size of 20 μm AED is 1 to 9%. In other words, the uncertainty associated with the performance characteristics of a $PM_{10}$ sampler sampling 10 μm particles is +/- 6% and +/- 4% when sampling 20 μm particles. These ranges are considered one form of inherent error associated with $PM_{10}$ ambient air samplers.

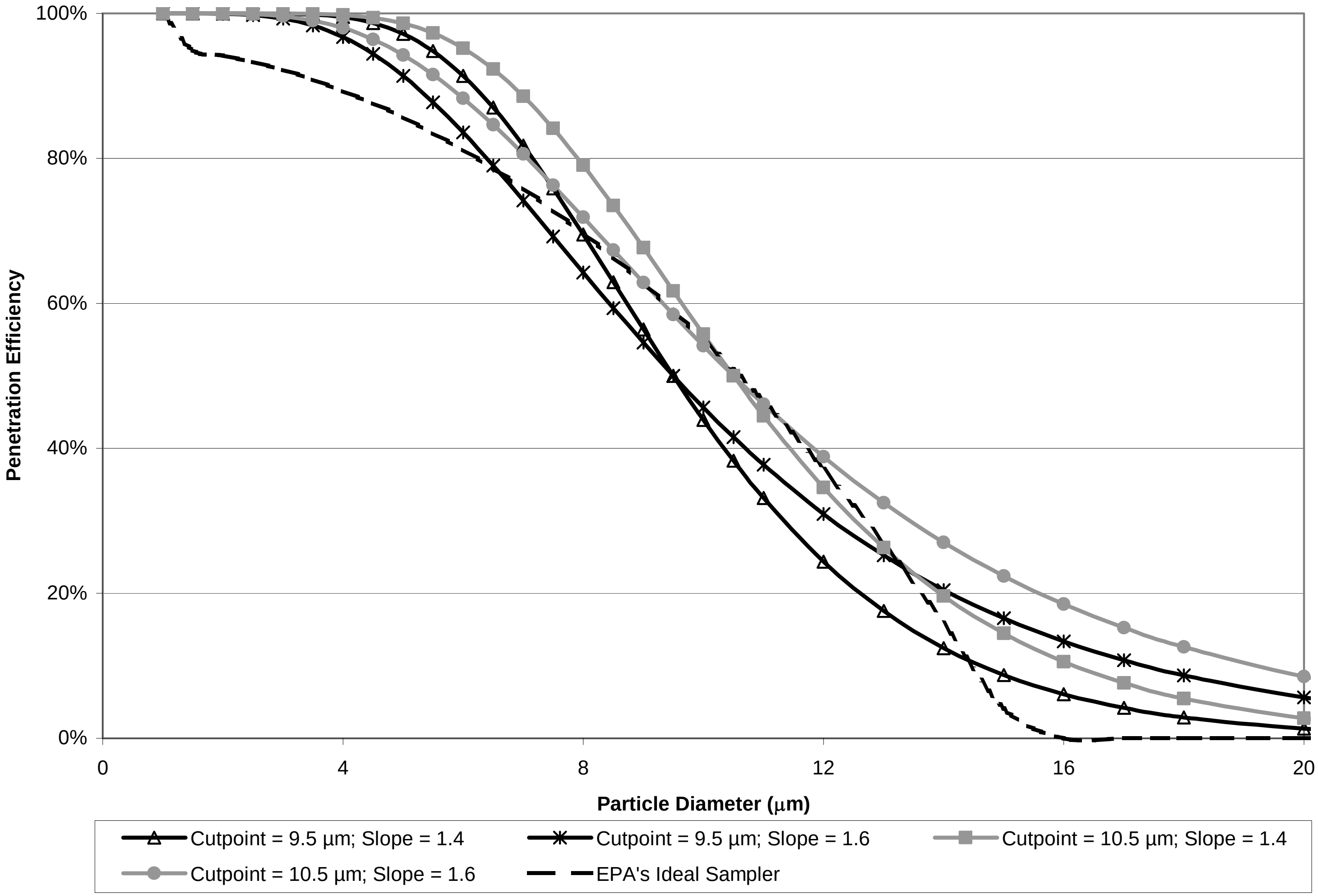
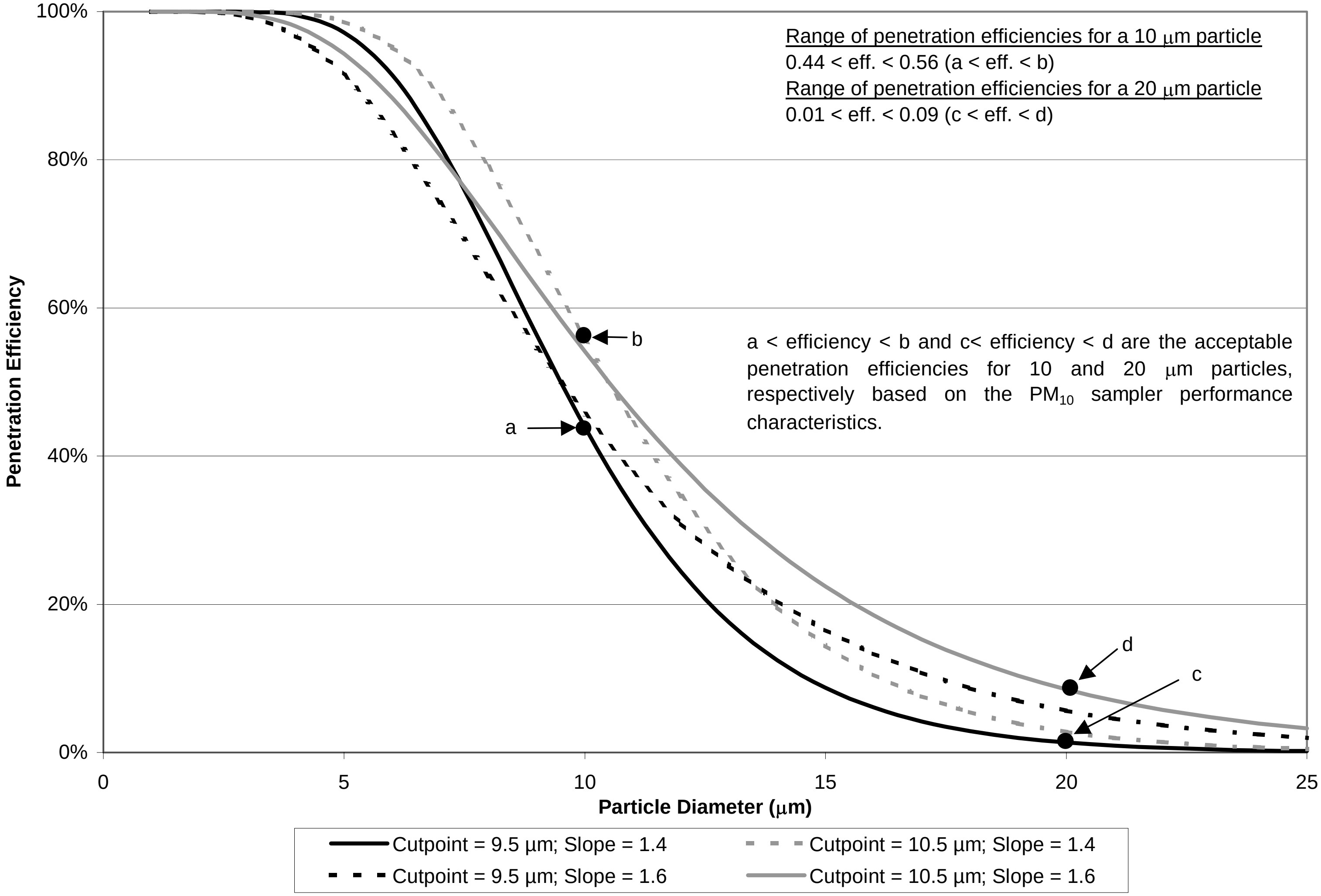


**Figure 17.** Comparison of the EPA (*CFR*, 2001e) ideal $PM_{10}$ sampler penetration data to the $PM_{10}$ sampler performance characteristics defined by Hinds (1982).



Range of penetration efficiencies for a 10 μm particle
0.44 < eff. < 0.56 (a < eff. < b)
Range of penetration efficiencies for a 20 μm particle
0.01 < eff. < 0.09 (c < eff. < d)

a < efficiency < b and c< efficiency < d are the acceptable penetration efficiencies for 10 and 20 μm particles, respectively based on the $PM_{10}$ sampler performance characteristics.

**Figure 18.** $PM_{10}$ sampler penetration curves based on the defining performance characteristics.

## Ambient $PM_{2.5}$ Sampler Performance Characteristics

According to the literature, EPA's emphasis on the 2.5 μm cutpoint was more closely associated with separating the fine and coarse atmospheric aerosol modes than mimicking a respiratory deposition convention. This emphasis is apparent when the penetration curve associated with the $PM_{2.5}$ ambient air sampler is compared to the ACGIH respirable fraction of PM, as shown in Figure 17.

EPA essentially defines the $d_{50}$ and slope associated with the $PM_{2.5}$ ambient air sampler in 40 CRF, Part 53 in the discussion of tests required for a Class II candidate sampler to receive EPA approval. The $d_{50}$ for the $PM_{2.5}$ sampler is explicitly stated in the EPA standards as 2.5 +/- 0.2 μm AED. No slope values for the sampler are listed in 40 CFR, Part 53 or any other current EPA standard; however, penetration data is presented in 40 CFR, Part 53. Ideally, the penetration data could be fit to a cumulative lognormal distribution to determine the characteristic $d_{50}$ and slope for the sampler; however, it was found that no single cumulative lognormal curve adequately represented the EPA data sets in 40 CFR, Part 53. It should be noted that this penetration data, along with EPA defined interval mass concentrations and mass penetration tolerances, are used to determine if a Class II sampler meets EPA's $PM_{2.5}$ performance criteria.

EPA's $PM_{2.5}$ cumulative penetration data set for Class II $PM_{2.5}$ candidate samplers produced a relatively smooth curve; however, the curve appeared to have a larger slope associated with particle sizes less than 2.5 μm AED than the slope associated with the particle sizes larger than 2.5 μm AED. It appears from the literature, that EPA intended for the $PM_{2.5}$ sampler to have a "sharp cut" or represent a true concentration of $PM_{2.5}$ which would mean that, ideally, the slope would be equal to 1.0. Work by Peters and Vanderpool (1996) suggested that a slope of 1.18 could be achieved with the WINS Impactor, an EPA approved sampler. Further work by Buch (1999) suggested that the slopes were not as sharp as previously reported and that a more appropriate estimation of the sampler slopes would be 1.32 +/- 0.03. For the purposes of this research, the $PM_{2.5}$ sampler performance characteristics will be defined as having a $d_{50}$ of 2.5 +/- 0.2 μm and a slope of 1.3 +/- 0.03.

The performance characteristic ranges used to define the ambient $PM_{2.5}$ sampler performance characteristics in this research were divided into nine $d_{50}$ and slope combinations: all combinations for $d_{50}$ values of 2.3, 2.5, and 2.7 μm and slope values of 1.27, 1.30, and 1.33. These nine sampler performance criteria were evaluated using EPA's wind tunnel evaluation guidelines for the ambient $PM_{2.5}$ sampler. The procedure is the same as that used in evaluating the ambient $PM_{10}$ sampler. The calculation values used in determining the expected mass concentration for a $PM_{2.5}$ sampler with a $d_{50}$ of 2.5 μm and a slope of 1.3 are shown in Tables 15-17 for the EPA defined idealized coarse aerosol, idealized "typical" coarse aerosol, and idealized fine coarse aerosol size distributions, respectively. According to 40 CFR, Part 53, a candidate sampler passes the sampling effectiveness test if the expected mass concentration calculated for the candidate sampler differs by no more than +/- 5% from that predicted for the ideal sampler (*CFR*, 2001d). The results of the comparison on the nine sampler performance criteria used in this research to that of EPA's ideal sampler are shown in Table 18. All the penetration curves evaluated passed the sampler effectiveness tests for the typical coarse and fine coarse aerosol size distributions; however, not all curves passed the test using coarse aerosol size distribution. The penetration curve defined by a $d_{50}$ of 2.5 μm and a slope of 1.33 and all curves defined with a $d_{50}$ of 2.7 μm failed the sampler effectiveness test for the coarse aerosol size distribution. Although some of the penetration curves generated from $d_{50}$ values of 2.5 +/- 0.2 μm and slope values of 1.3 +/- 0.03 failed the sampler effectiveness tests, these performance criteria ranges were used throughout the remainder of this research since these ranges have been observed in the actual evaluation of EPA approved $PM_{2.5}$ samplers.

**Table 15.** Expected mass concentration for a $PM_{2.5}$ sampler with a cutpoint of 2.5 μm and a slope of 1.3 and the EPA ideal $PM_{2.5}$ sampler in accordance with the EPA wind tunnel evaluation guidelines for an idealized coarse aerosol size distribution (*CFR*, 2001d).

| | Test Sampler | | | Ideal Sampler | | |
|---|---|---|---|---|---|---|
| Particle Size (μm) | Sampling Effectiveness | Interval Mass Conc. ($\mu g/m^3$) | Expected Mass Conc. ($\mu g/m^3$) | Sampling Effectiveness | Interval Mass Conc. ($\mu g/m^3$) | Expected Mass Conc. ($\mu g/m^3$) |
| < 0.500 | 1.000 | 6.001 | 6.001 | 1.000 | 6.001 | 6.001 |
| 0.625 | 1.000 | 2.129 | 2.129 | 0.999 | 2.129 | 2.127 |
| 0.750 | 1.000 | 0.982 | 0.982 | 0.998 | 0.982 | 0.980 |
| 0.875 | 1.000 | 0.730 | 0.730 | 0.997 | 0.730 | 0.728 |
| 1.000 | 1.000 | 0.551 | 0.551 | 0.995 | 0.551 | 0.548 |
| 1.125 | 0.999 | 0.428 | 0.427 | 0.991 | 0.428 | 0.424 |
| 1.250 | 0.996 | 0.346 | 0.345 | 0.987 | 0.346 | 0.342 |
| 1.375 | 0.989 | 0.294 | 0.291 | 0.980 | 0.294 | 0.288 |
| 1.500 | 0.974 | 0.264 | 0.257 | 0.969 | 0.264 | 0.256 |
| 1.675 | 0.937 | 0.251 | 0.235 | 0.954 | 0.251 | 0.239 |
| 1.750 | 0.913 | 0.250 | 0.228 | 0.932 | 0.250 | 0.233 |
| 1.875 | 0.864 | 0.258 | 0.223 | 0.899 | 0.258 | 0.232 |
| 2.000 | 0.802 | 0.272 | 0.218 | 0.854 | 0.272 | 0.232 |
| 2.125 | 0.732 | 0.292 | 0.214 | 0.791 | 0.292 | 0.231 |
| 2.250 | 0.656 | 0.314 | 0.206 | 0.707 | 0.314 | 0.222 |
| 2.375 | 0.578 | 0.339 | 0.196 | 0.602 | 0.339 | 0.204 |
| 2.500 | 0.500 | 0.366 | 0.183 | 0.480 | 0.366 | 0.176 |
| 2.625 | 0.426 | 0.394 | 0.168 | 0.351 | 0.394 | 0.138 |
| 2.750 | 0.358 | 0.422 | 0.151 | 0.230 | 0.422 | 0.097 |
| 2.875 | 0.297 | 0.449 | 0.133 | 0.133 | 0.449 | 0.060 |
| 3.000 | 0.244 | 0.477 | 0.116 | 0.067 | 0.477 | 0.032 |
| 3.125 | 0.198 | 0.504 | 0.100 | 0.030 | 0.504 | 0.015 |
| 3.250 | 0.159 | 0.530 | 0.084 | 0.012 | 0.530 | 0.006 |
| 3.375 | 0.126 | 0.555 | 0.070 | 0.004 | 0.555 | 0.002 |
| 3.500 | 0.100 | 0.579 | 0.058 | 0.001 | 0.579 | 0.001 |
| 3.625 | 0.078 | 0.602 | 0.047 | 0.000 | 0.602 | 0.000 |
| 3.750 | 0.061 | 0.624 | 0.038 | 0.000 | 0.624 | 0.000 |
| 3.975 | 0.039 | 0.644 | 0.025 | 0.000 | 0.644 | 0.000 |
| 4.000 | 0.037 | 0.663 | 0.024 | 0.000 | 0.663 | 0.000 |
| 4.125 | 0.028 | 0.681 | 0.019 | 0.000 | 0.681 | 0.000 |
| 4.250 | 0.022 | 0.697 | 0.015 | 0.000 | 0.697 | 0.000 |
| 4.375 | 0.016 | 0.712 | 0.012 | 0.000 | 0.712 | 0.000 |
| 4.500 | 0.013 | 0.726 | 0.009 | 0.000 | 0.726 | 0.000 |
| 4.625 | 0.010 | 0.738 | 0.007 | 0.000 | 0.738 | 0.000 |
| 4.750 | 0.007 | 0.750 | 0.005 | 0.000 | 0.750 | 0.000 |
| 4.875 | 0.005 | 0.760 | 0.004 | 0.000 | 0.760 | 0.000 |
| 5.000 | 0.004 | 0.769 | 0.003 | 0.000 | 0.769 | 0.000 |
| 5.125 | 0.003 | 0.777 | 0.002 | 0.000 | 0.777 | 0.000 |
| 5.250 | 0.002 | 0.783 | 0.002 | 0.000 | 0.783 | 0.000 |
| 5.375 | 0.002 | 0.789 | 0.001 | 0.000 | 0.789 | 0.000 |
| 5.500 | 0.001 | 0.794 | 0.001 | 0.000 | 0.794 | 0.000 |
| 5.625 | 0.001 | 0.798 | 0.001 | 0.000 | 0.798 | 0.000 |
| 5.750 | 0.001 | 0.801 | 0.001 | 0.000 | 0.801 | 0.000 |
| | | $C_{sam(exp)}$ | 14.513 | | $C_{ideal(exp)}$ | 13.814 |

**Table 16.** Expected mass concentration for a $PM_{2.5}$ sampler with a cutpoint of 2.5 μm and a slope of 1.3 and the EPA ideal $PM_{2.5}$ sampler in accordance with the EPA wind tunnel evaluation guidelines for an idealized "typical" coarse aerosol size distribution (*CFR*, 2001d).

| | Test Sampler | | | Ideal Sampler | | |
|---|---|---|---|---|---|---|
| Particle Size (μm) | Sampling Effectiveness | Interval Mass Conc. (μg/$m^3$) | Expected Mass Conc. (μg/$m^3$) | Sampling Effectiveness | Interval Mass Conc. (μg/$m^3$) | Expected Mass Conc. (μg/$m^3$) |
| <0.5 | 1.000 | 16.651 | 16.651 | 1.000 | 16.651 | 16.651 |
| 0.625 | 1.000 | 5.899 | 5.899 | 0.999 | 5.899 | 5.893 |
| 0.750 | 1.000 | 2.708 | 2.708 | 0.998 | 2.708 | 2.703 |
| 0.875 | 1.000 | 1.996 | 1.996 | 0.997 | 1.996 | 1.990 |
| 1.000 | 1.000 | 1.478 | 1.478 | 0.995 | 1.478 | 1.471 |
| 1.125 | 0.999 | 1.108 | 1.107 | 0.991 | 1.108 | 1.098 |
| 1.250 | 0.996 | 0.846 | 0.843 | 0.987 | 0.846 | 0.835 |
| 1.375 | 0.989 | 0.661 | 0.654 | 0.980 | 0.661 | 0.648 |
| 1.500 | 0.974 | 0.532 | 0.518 | 0.969 | 0.532 | 0.516 |
| 1.675 | 0.937 | 0.444 | 0.416 | 0.954 | 0.444 | 0.424 |
| 1.750 | 0.913 | 0.384 | 0.351 | 0.932 | 0.384 | 0.358 |
| 1.875 | 0.864 | 0.347 | 0.300 | 0.899 | 0.347 | 0.312 |
| 2.000 | 0.802 | 0.325 | 0.261 | 0.854 | 0.325 | 0.278 |
| 2.125 | 0.732 | 0.314 | 0.230 | 0.791 | 0.314 | 0.248 |
| 2.250 | 0.656 | 0.312 | 0.205 | 0.707 | 0.312 | 0.221 |
| 2.375 | 0.578 | 0.316 | 0.182 | 0.602 | 0.316 | 0.190 |
| 2.500 | 0.500 | 0.325 | 0.163 | 0.480 | 0.325 | 0.156 |
| 2.625 | 0.426 | 0.336 | 0.143 | 0.351 | 0.336 | 0.118 |
| 2.750 | 0.358 | 0.350 | 0.125 | 0.230 | 0.350 | 0.081 |
| 2.875 | 0.297 | 0.366 | 0.109 | 0.133 | 0.366 | 0.049 |
| 3.000 | 0.244 | 0.382 | 0.093 | 0.067 | 0.382 | 0.026 |
| 3.125 | 0.198 | 0.399 | 0.079 | 0.030 | 0.399 | 0.012 |
| 3.250 | 0.159 | 0.416 | 0.066 | 0.012 | 0.416 | 0.005 |
| 3.375 | 0.126 | 0.432 | 0.055 | 0.004 | 0.432 | 0.002 |
| 3.500 | 0.100 | 0.449 | 0.045 | 0.001 | 0.449 | 0.000 |
| 3.625 | 0.078 | 0.464 | 0.036 | 0.000 | 0.464 | 0.000 |
| 3.750 | 0.061 | 0.480 | 0.029 | 0.000 | 0.480 | 0.000 |
| 3.975 | 0.039 | 0.494 | 0.019 | 0.000 | 0.494 | 0.000 |
| 4.000 | 0.037 | 0.507 | 0.019 | 0.000 | 0.507 | 0.000 |
| 4.125 | 0.028 | 0.520 | 0.015 | 0.000 | 0.520 | 0.000 |
| 4.250 | 0.022 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 4.375 | 0.016 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 4.500 | 0.013 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 4.625 | 0.010 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 4.750 | 0.007 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 4.875 | 0.005 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 5.000 | 0.004 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 5.125 | 0.003 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 5.250 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 5.375 | 0.002 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 5.500 | 0.001 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 5.625 | 0.001 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 5.750 | 0.001 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | | $C_{sam(exp)}$) | 34.792 | | $C_{ideal(exp)}$ | 34.282 |

**Table 17.** Expected mass concentration for a $PM_{2.5}$ sampler with a cutpoint of 2.5 μm and a slope of 1.3 and the EPA ideal $PM_{2.5}$ sampler in accordance with the EPA wind tunnel evaluation guidelines for an idealized fine coarse aerosol size distribution (*CFR*, 2001d).

| | Test Sampler | | | Ideal Sampler | | |
|---|---|---|---|---|---|---|
| Particle Size (μm) | Sampling Effectiveness | Interval Mass Conc. (μg/m$^3$) | Expected Mass Conc. (μg/m$^3$) | Sampling Effectiveness | Interval Mass Conc. (μg/m$^3$) | Expected Mass Conc. (μg/m$^3$) |
| <0.5 | 1.000 | 18.868 | 18.868 | 1.000 | 18.868 | 18.868 |
| 0.625 | 1.000 | 13.412 | 13.412 | 0.999 | 13.412 | 13.399 |
| 0.750 | 1.000 | 8.014 | 8.014 | 0.998 | 8.014 | 7.998 |
| 0.875 | 1.000 | 6.984 | 6.984 | 0.997 | 6.984 | 6.963 |
| 1.000 | 1.000 | 5.954 | 5.953 | 0.995 | 5.954 | 5.924 |
| 1.125 | 0.999 | 5.015 | 5.009 | 0.991 | 5.015 | 4.970 |
| 1.250 | 0.996 | 4.197 | 4.180 | 0.987 | 4.197 | 4.142 |
| 1.375 | 0.989 | 3.503 | 3.463 | 0.980 | 3.503 | 3.433 |
| 1.500 | 0.974 | 2.921 | 2.846 | 0.969 | 2.921 | 2.830 |
| 1.675 | 0.937 | 2.438 | 2.283 | 0.954 | 2.438 | 2.326 |
| 1.750 | 0.913 | 2.039 | 1.862 | 0.932 | 2.039 | 1.900 |
| 1.875 | 0.864 | 1.709 | 1.476 | 0.899 | 1.709 | 1.536 |
| 2.000 | 0.802 | 1.437 | 1.153 | 0.854 | 1.437 | 1.227 |
| 2.125 | 0.732 | 1.212 | 0.887 | 0.791 | 1.212 | 0.959 |
| 2.250 | 0.656 | 1.026 | 0.673 | 0.707 | 1.026 | 0.725 |
| 2.375 | 0.578 | 0.873 | 0.504 | 0.602 | 0.873 | 0.526 |
| 2.500 | 0.500 | 0.745 | 0.373 | 0.480 | 0.745 | 0.358 |
| 2.625 | 0.426 | 0.638 | 0.272 | 0.351 | 0.638 | 0.224 |
| 2.750 | 0.358 | 0.550 | 0.197 | 0.230 | 0.550 | 0.127 |
| 2.875 | 0.297 | 0.476 | 0.141 | 0.133 | 0.476 | 0.063 |
| 3.000 | 0.244 | 0.414 | 0.101 | 0.067 | 0.414 | 0.028 |
| 3.125 | 0.198 | 0.362 | 0.072 | 0.030 | 0.362 | 0.011 |
| 3.250 | 0.159 | 0.319 | 0.051 | 0.012 | 0.319 | 0.004 |
| 3.375 | 0.126 | 0.282 | 0.036 | 0.004 | 0.282 | 0.001 |
| 3.500 | 0.100 | 0.252 | 0.025 | 0.001 | 0.252 | 0.000 |
| 3.625 | 0.078 | 0.226 | 0.018 | 0.000 | 0.226 | 0.000 |
| 3.750 | 0.061 | 0.204 | 0.012 | 0.000 | 0.204 | 0.000 |
| 3.975 | 0.039 | 0.185 | 0.007 | 0.000 | 0.185 | 0.000 |
| 4.000 | 0.037 | 0.170 | 0.006 | 0.000 | 0.170 | 0.000 |
| 4.125 | 0.028 | 0.157 | 0.004 | 0.000 | 0.157 | 0.000 |
| 4.250 | 0.022 | 0.146 | 0.003 | 0.000 | 0.146 | 0.000 |
| 4.375 | 0.016 | 0.136 | 0.002 | 0.000 | 0.136 | 0.000 |
| 4.500 | 0.013 | 0.129 | 0.002 | 0.000 | 0.129 | 0.000 |
| 4.625 | 0.010 | 0.122 | 0.001 | 0.000 | 0.122 | 0.000 |
| 4.750 | 0.007 | 0.117 | 0.001 | 0.000 | 0.117 | 0.000 |
| 4.875 | 0.005 | 0.112 | 0.001 | 0.000 | 0.112 | 0.000 |
| 5.000 | 0.004 | 0.108 | 0.000 | 0.000 | 0.108 | 0.000 |
| 5.125 | 0.003 | 0.105 | 0.000 | 0.000 | 0.105 | 0.000 |
| 5.250 | 0.002 | 0.102 | 0.000 | 0.000 | 0.102 | 0.000 |
| 5.375 | 0.002 | 0.100 | 0.000 | 0.000 | 0.100 | 0.000 |
| 5.500 | 0.001 | 0.098 | 0.000 | 0.000 | 0.098 | 0.000 |
| 5.625 | 0.001 | 0.097 | 0.000 | 0.000 | 0.097 | 0.000 |
| 5.750 | 0.001 | 0.096 | 0.000 | 0.000 | 0.096 | 0.000 |
| | | $C_{sam(exp)}$) | 78.892 | | $C_{ideal(exp)}$ | 78.542 |

**Table 18.** Estimated $PM_{2.5}$ mass concentration ratios between sampler performance characteristics and the EPA idealized sampler.

| Coarse Aerosol | | | "Typical" Coarse Aerosol | | | Fine Aerosol | | |
|---|---|---|---|---|---|---|---|---|
| Cutpoint (μm) | Slope | Ratio (%) | Cutpoint (μm) | Slope | Ratio (%) | Cutpoint (μm) | Slope | Ratio (%) |
| 2.3 | 1.27 | 100 | 2.3 | 1.27 | 100 | 2.3 | 1.27 | 99 |
| 2.3 | 1.30 | 100 | 2.3 | 1.30 | 100 | 2.3 | 1.30 | 99 |
| 2.3 | 1.33 | 101 | 2.3 | 1.33 | 100 | 2.3 | 1.33 | 98 |
| 2.5 | 1.27 | 104 | 2.5 | 1.27 | 101 | 2.5 | 1.27 | 101 |
| 2.5 | 1.30 | 105 | 2.5 | 1.30 | 101 | 2.5 | 1.30 | 100 |
| 2.5 | 1.33 | 106 | 2.5 | 1.33 | 102 | 2.5 | 1.33 | 100 |
| 2.7 | 1.27 | 109 | 2.7 | 1.27 | 103 | 2.7 | 1.27 | 102 |
| 2.7 | 1.30 | 110 | 2.7 | 1.30 | 103 | 2.7 | 1.30 | 102 |
| 2.7 | 1.33 | 111 | 2.7 | 1.33 | 103 | 2.7 | 1.33 | 102 |

Based on the ambient $PM_{2.5}$ sampler performance criteria used in this research, four combinations of $d_{50}$ and slope values were used to define boundary penetration efficiency curves. These penetration curves were defined with $d_{50}$ values of 2.3 and 2.7 μm and slope values of 1.27 and 1.33. Figure 19 illustrates the comparison of the boundary penetration curves used in this research and the EPA ideal $PM_{2.5}$ sampler penetration efficiency curve. The ideal penetration curve is encompassed by the boundary penetration curves for particle diameters less than about 2.7 μm and outside of this range for particle diameters greater than 2.7 μm.

When comparing the boundary penetration efficiency curves in Figure 20, it is apparent that there is an acceptable range of penetration efficiencies for the $PM_{2.5}$ ambient air sampler. The acceptable range of penetration efficiencies for a particle size of 2.5 μm AED is 36 to 63%. In other words, the uncertainty associated with the performance characteristics of a $PM_{2.5}$ ambient air sampler when sampling 2.5 μm particles is +/- 16.5%. These ranges are considered one form of inherent error associated with $PM_{2.5}$ ambient air samplers.

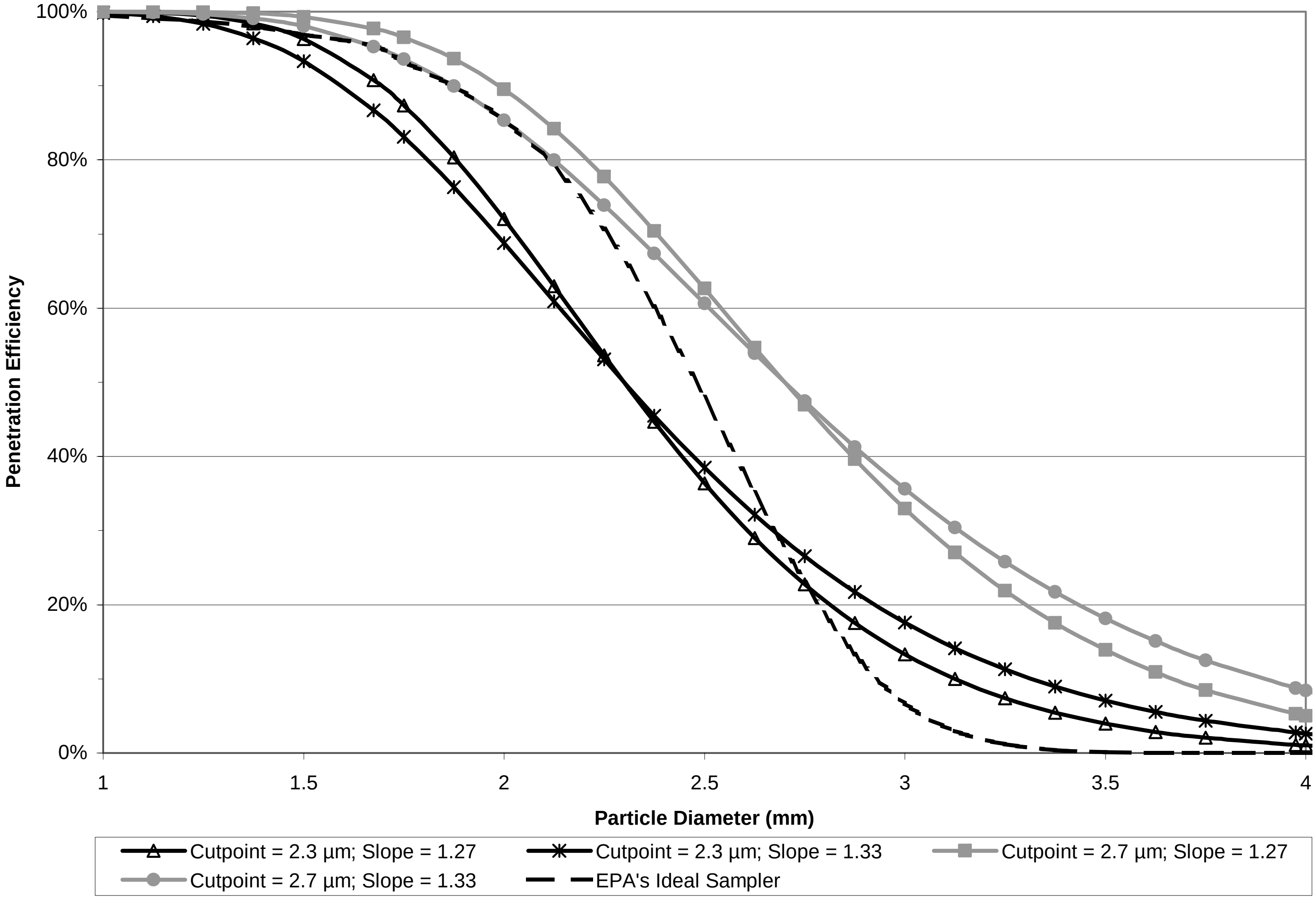


**Figure 19.** Comparison of the EPA (*CFR*, 2001d) ideal $PM_{2.5}$ sampler penetration data to the $PM_{2.5}$ sampler performance characteristics used in this research.

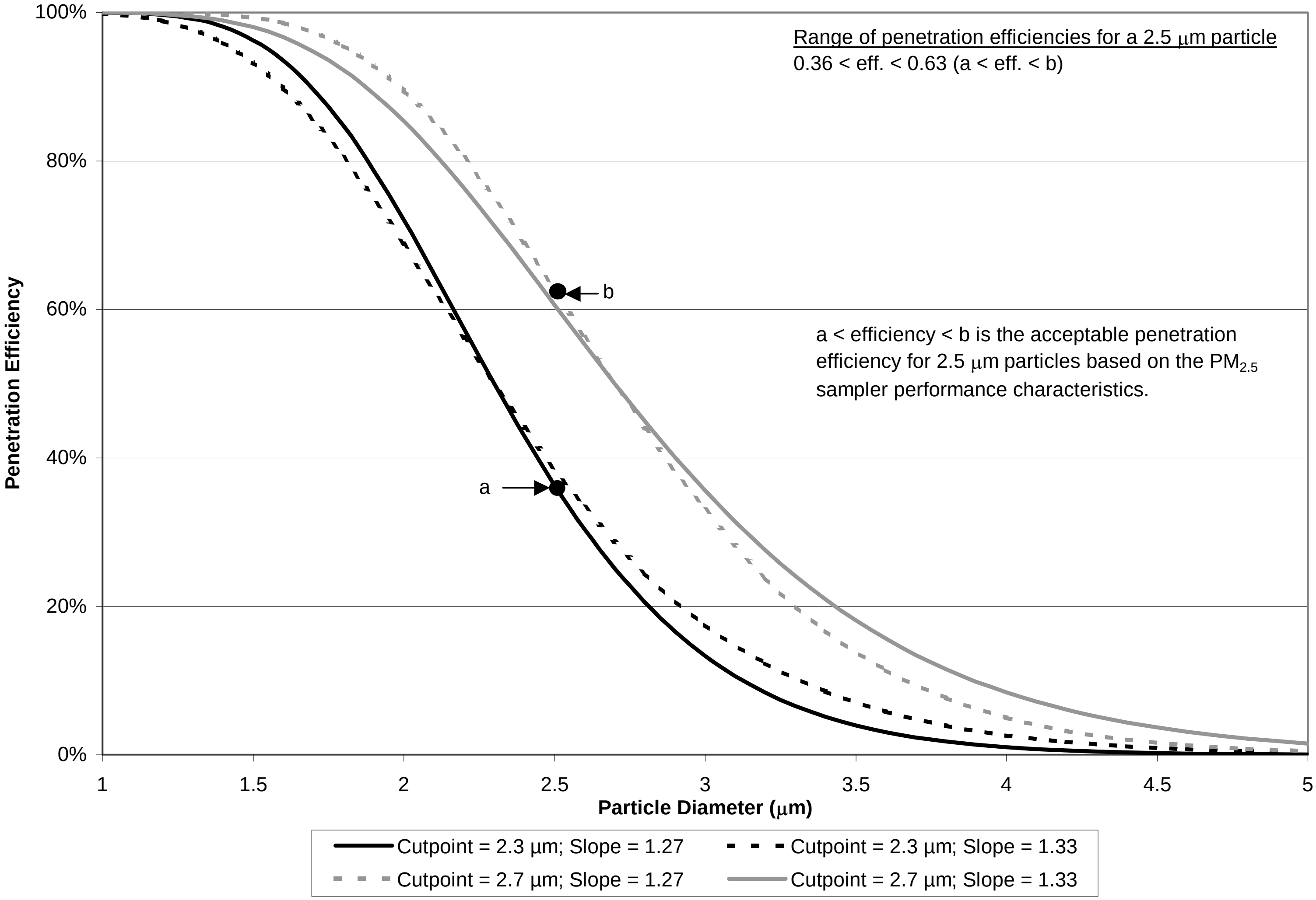


**Figure 20.** $PM_{2.5}$ sampler penetration curves based on the defining performance characteristics.

## $PM_{10}$ Stack Sampler Performance Characteristics

Stack samplers approved by EPA are designed quite differently than EPA approved ambient air samplers, as one would expect. However, the details provided by EPA regarding $PM_{10}$ stack samplers are very limited in comparison to ambient air samplers. In addition, it would seem logical that EPA would impose the same sampler performance standards on stack samplers as imposed for ambient $PM_{10}$ samplers. The limited information provide by EPA on the stack sampler performance characteristics includes a target cutpoint and corresponding tolerances and an acceptable collection efficiency envelope.

The $d_{50}$ for the $PM_{10}$ stack sampler is explicitly defined as 10.0 +/-1.0 μm AED. Equation 16 with $a$ equal to infinity was used to determine slope values associated with $d_{50}$'s of 9.0, 10.0, and 11.0 μm that fall within the EPA acceptable Method 201a $PM_{10}$ cyclone efficiency envelope. The resulting collection efficiency curves are shown in Figure 21 along with the EPA collection efficiency envelope. It was determined that collection efficiency curve of $PM_{10}$ stack sampling cyclone could be described by a $d_{50}$ of 9 μm and a slope of 1.87, a $d_{50}$ of 10 μm and a slope of 1.90, or a $d_{50}$ of 11 μm and a slope of 1.76 and still meet the EPA performance requirements. Based on these performance criteria, 12 to 23% of 5.7 μm, 43 to 57% of the 10 μm, and 85 to 90% of the 20 μm particles could be captured by the Method 201a $PM_{10}$ sampler cyclone. Based on the EPA efficiency envelope, the Method 201a $PM_{10}$ sampler cyclone could capture 100% of the particles greater than 10 μm and allow 100% of the particle less than 10 μm to pass to the filter and still considered an acceptable sampler. In other words, a Method 201a sampling cyclone could capture 0 to 23% of the 5.7 μm, 0 to 57% of the 10 μm, and 85 to 100% of the 20 μm particles and still fall within the EPA performance criteria. In other words, the performance characteristic uncertainty associated with the EPA $PM_{10}$ stack sampler is +/- 11.5%, +/- 26%, and +/- 7.5% when sampling 5.7, 10.0, and 20.0 μm particles, respectively.

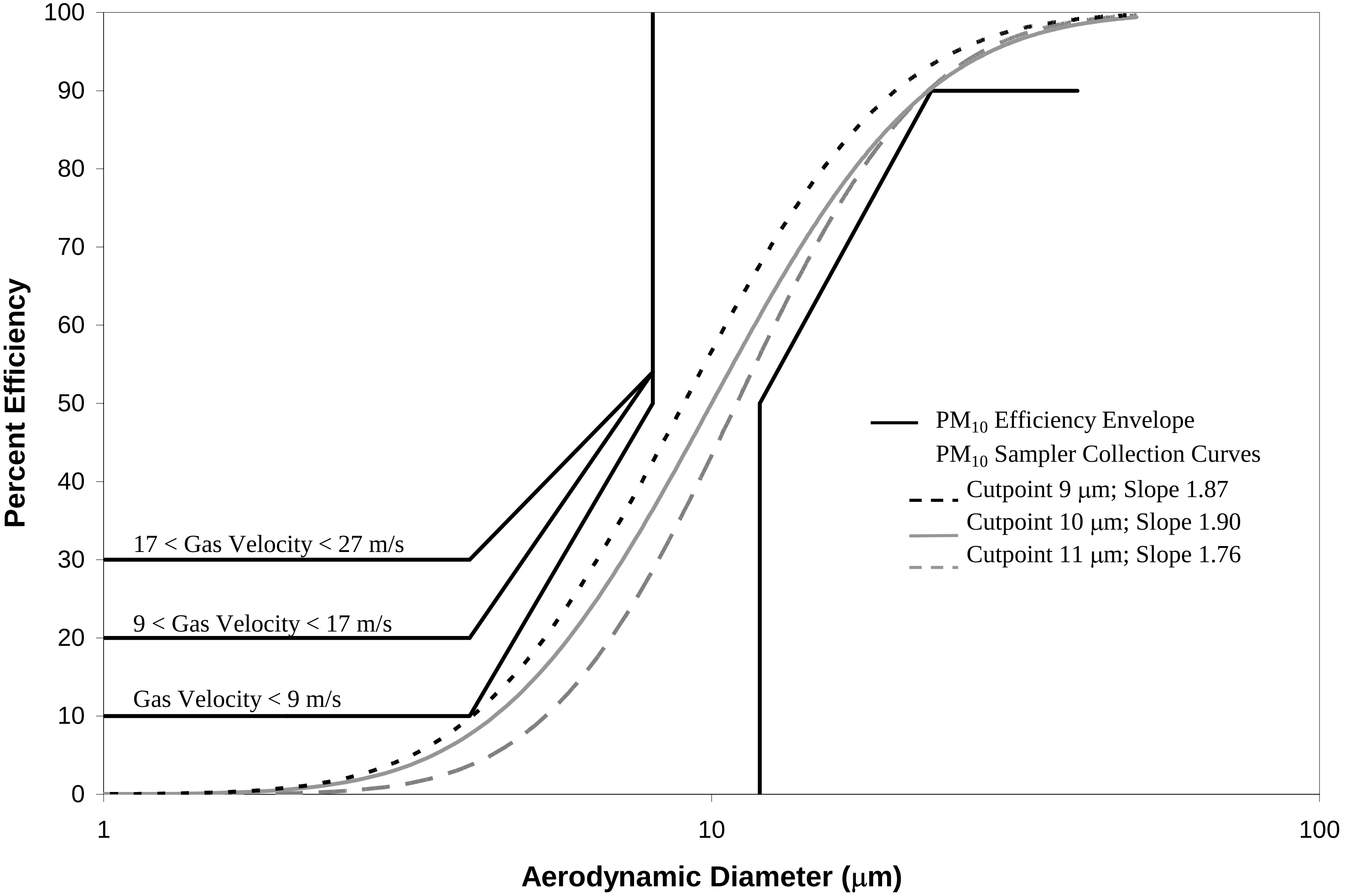


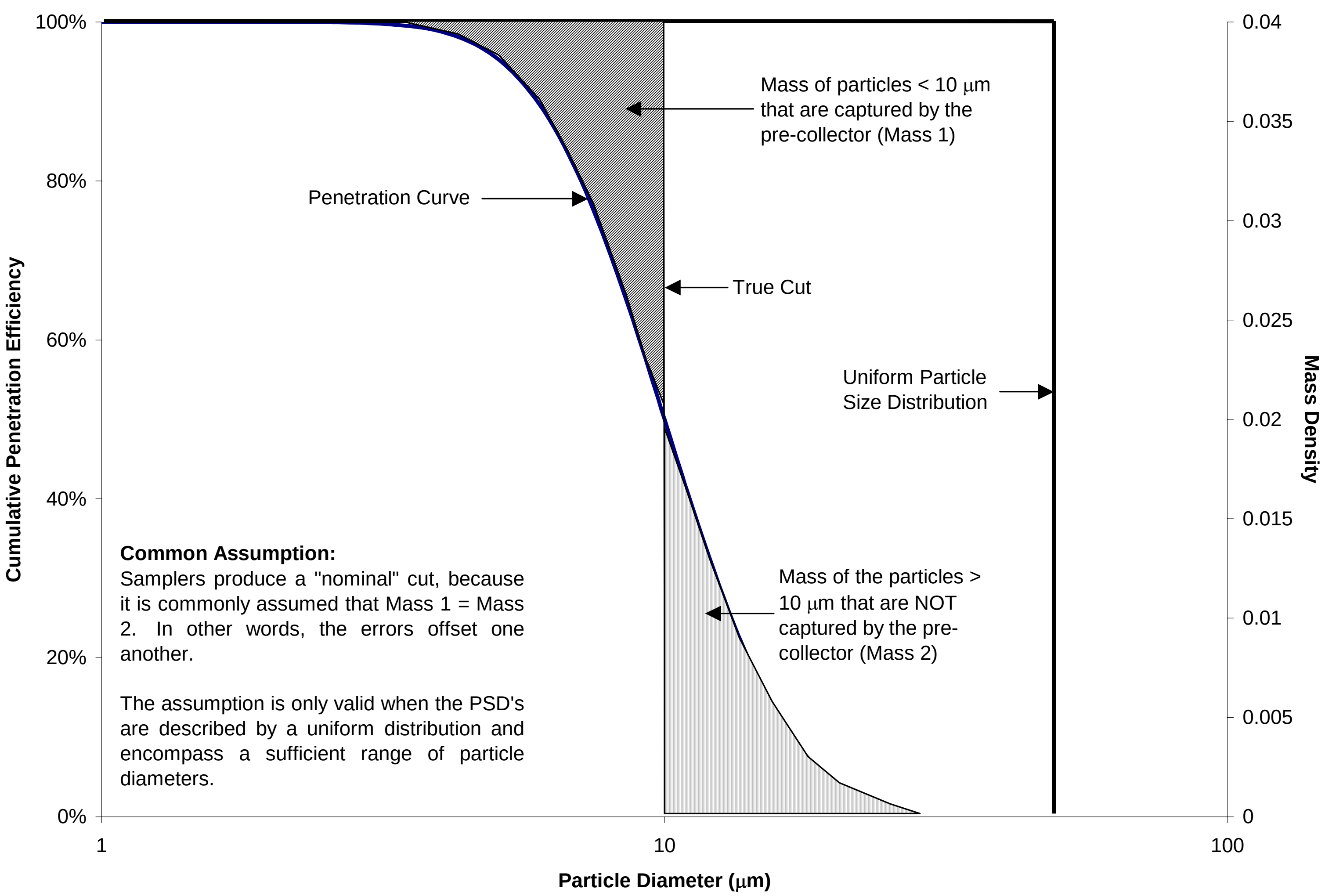
**Figure 21.** Method 201a $PM_{10}$ cyclone efficiency envelope and theoretical $PM_{10}$ cyclone collection efficiency curves.

## Interaction of Particle Size and Sampler Performance Characteristics

In determining the errors associated with PM samplers, the most important question to ask is “what is the intent of the PM regulations?”. It was previously established that the primary purpose of the regulations is to protect public health. It is quite clear in the literature that PM collected from a $PM_{10}$ ambient air sampler should mimic the fraction of PM that penetrates to the thoracic region of the human respiratory system. An assumption made in the $PM_{10}$ regulation is that it pertains to a measure of particles with an AED less than or equal to a nominal 10 μm. Unlike the published intent of the $PM_{10}$ ambient air sampler, the literature indicates that EPA intended for the $PM_{2.5}$ sampler to be a true measure of PM with particle diameters less than or equal to 2.5 μm. An assumption made in the $PM_{2.5}$ regulation is that it pertains to a measure of particles with an AED less than or equal to a nominal 2.5 μm. The term nominal implies that the measured PM described in the $PM_{10}$ ($PM_{2.5}$) regulation does not account for all mass associated with particles less than or equal 10 μm (2.5 μm) AED and does include some of the mass associated with particles larger than 10 μm (2.5 μm) AED.

This issue of nominal values leads to a primary focus of this section, which is industries that emit PM with a MMD less than or equal to 5.7 μm (MMD associated with the EPA definition of an urban dust) are not regulated at the same level as agricultural operations that typically emit PM with a MMD much greater than 5.7 μm. This unequal regulation is primarily because of the interaction of the sampler performance and PSD characteristics.

Mathcad 2000 was used to evaluate equations 18 through 23 for various PSD and $PM_{10}$ sampler performance characteristics previously defined in order to obtain a general concept of how the interaction of these characteristics impacts the current regulation of $PM_{10}$. The results of the evaluation were graphically illustrated in Figures 22 through 26.



**Common Assumption:**
Samplers produce a "nominal" cut, because it is commonly assumed that Mass 1 = Mass 2. In other words, the errors offset one another.

The assumption is only valid when the PSD's are described by a uniform distribution and encompass a sufficient range of particle diameters.

**Figure 22.** Sampler nominal cut for a uniform PSD.

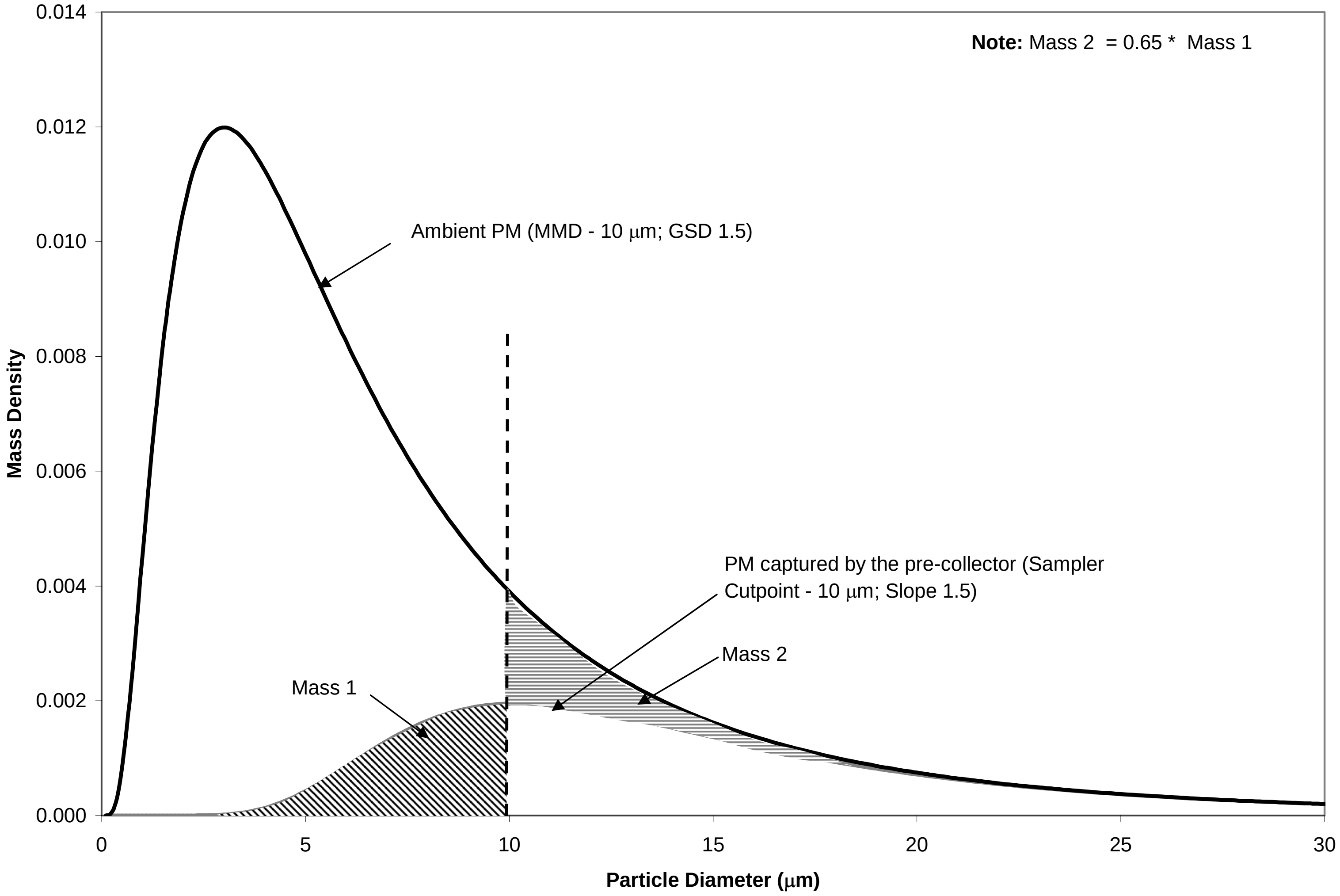


**Figure 23.** Sampler nominal cut for a lognormal PSD with a MMD = 5.7 μm and GSD = 2.25.

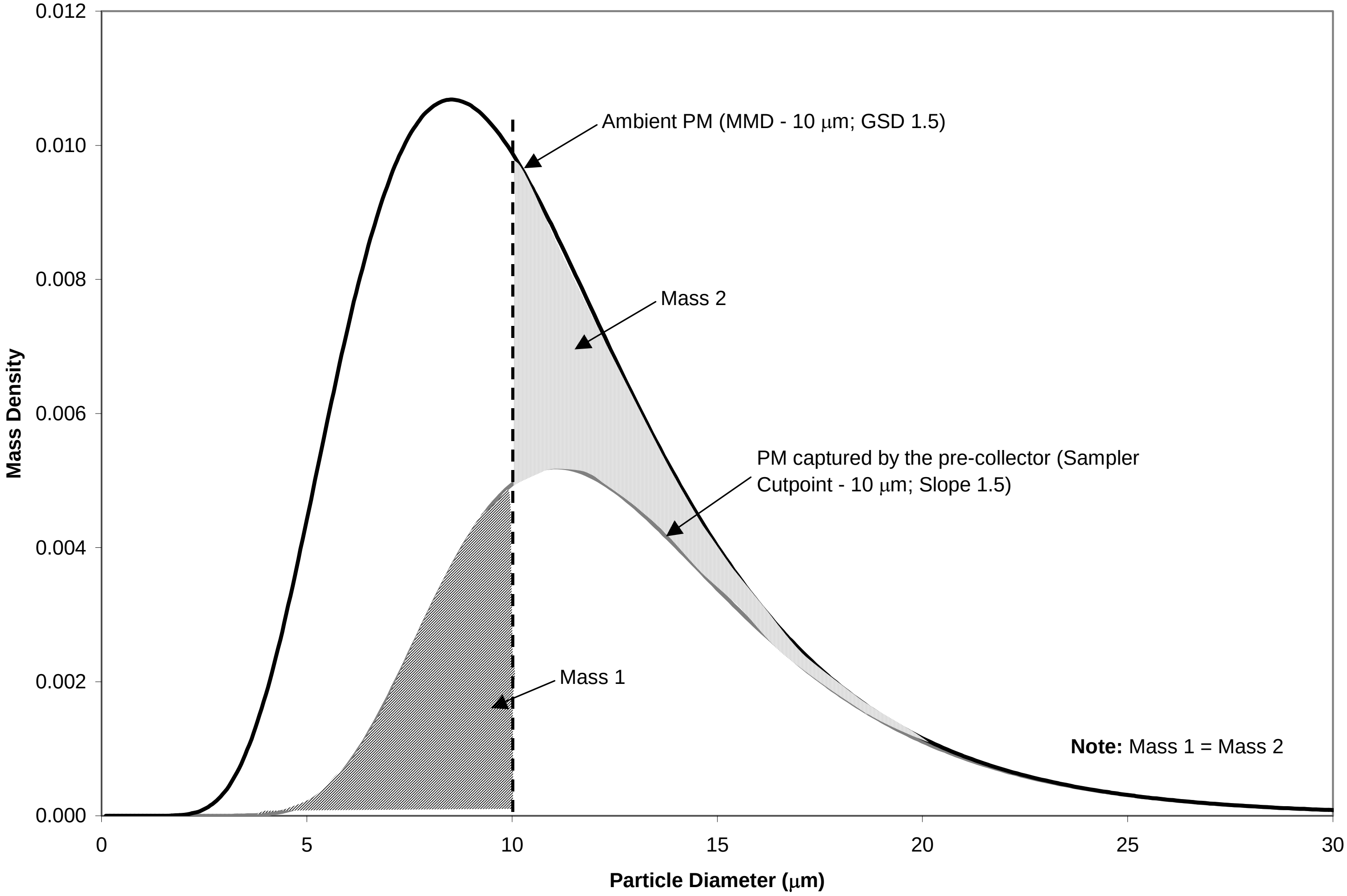


**Figure 24.** Sampler nominal cut for a lognormal PSD with a MMD = 10 μm and GSD = 1.5.

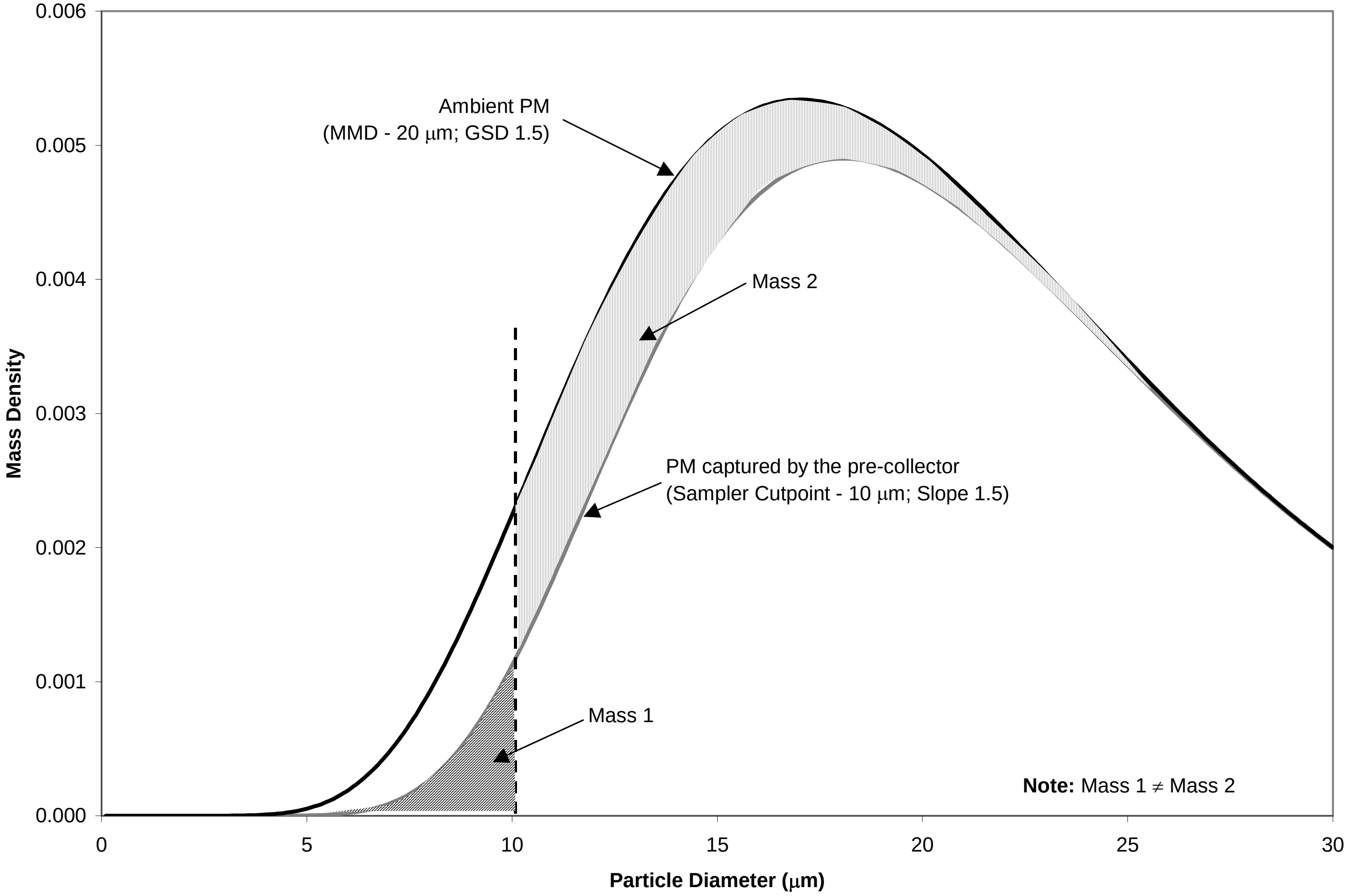


**Figure 25.** Sampler nominal cut for a lognormal PSD with a MMD = 20 µm and GSD = 1.5.

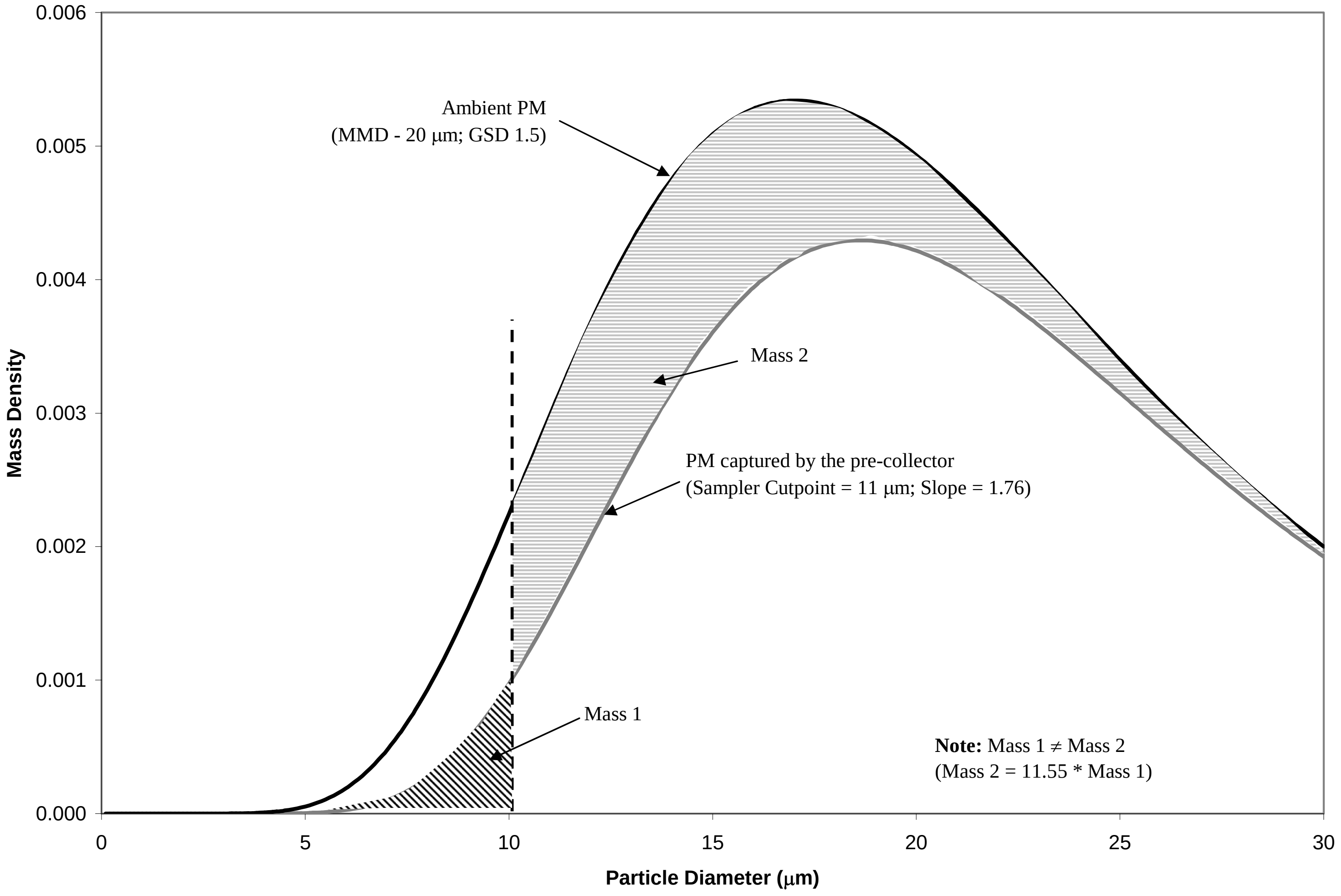


**Figure 26.** $PM_{10}$ stack sampler nominal cut (sampler $d_{50}$ = 11 μm; slope = 1.76) for a lognormal PSD with a MMD = 20 μm and GSD = 1.5.

Figure 22 illustrates the interaction associated with a uniformly distributed PSD and the optimum ambient $PM_{10}$ sampler performance characteristics. The two errors associated with the interaction are highlighted in the figure and labeled as mass 1 and mass 2. Mass 1 refers to an under-sampling error, while mass 2 corresponds to an over-sampling error. A common assumption made in the regulatory community to circumvent the problem associated with the two errors is that the mass of particles less than 10 μm and captured by the pre-separator (mass 1) is equal to the mass of particle greater than 10 μm and captured on the filter (mass 2). This assumption is valid when the density function of the PSD of the dust in the air being sampled is represented by a uniform distribution, i.e., mass 1 equals mass 2. When the under-sampling error is equal to the over-sampling error, the concentration measured by the sampler is equal to the true concentration.

The outermost curve in Figure 23 corresponds to a lognormal PSD with a MMD of 5.7 μm and a GSD of 2.25 (PSD characteristics associated with urban dust as defined by EPA). The area below the innermost curve represents the mass of particles captured by the $PM_{10}$ preseparator and the area between the two curves represents the mass of particles that penetrated the preseparator and were captured on the filter. In this situation, mass 2, or the mass of particles associated with particle diameters greater than 10 μm that penetrate the pre-separator and are captured on the filter, is less than mass 1, or the mass of particles with diameters less than 10 μm that are captured by the pre-separator and are not captured on the filter. When mass 1 is greater than mass 2, the $PM_{10}$ sampler is under-estimating the $PM_{10}$ concentration. For an urban PSD, the over-sampling error is 0.65 times the under-sampling error.

Figure 24 is similar to Figure 23, with the exception that the PSD characteristics are defined by a MMD of 10 μm and a GSD of 1.5. As in Figure 23, the two errors associated with the interaction of the PSD and sampler performance characteristics are highlighted in Figure 24 and labeled mass 1 and mass 2. In this situation where the MMD of the PSD is equal to the cutpoint of the preseparator, mass 1 is equal to mass 2 and the nominal assumption is valid and the sampler concentration is equal to the true concentration.

Figure 25 is similar to Figure 24, with the exception that the PSD characteristics are defined by a MMD of 20 μm and a GSD of 1.5. In this situation, mass 2, or the mass of particles associated with particle diameters greater than 10 μm that penetrate the pre-separator and are captured on the filter, is greater than mass 1, or the mass of particles with diameters less than 10 μm that are captured by the pre-separator and are not captured on the filter. For the given PSD characteristics, the concentration measured by this sampler is approximately 3.4 times the true $PM_{10}$ concentration. When mass 1 is greater than mass 2, the $PM_{10}$ sampler is over-estimating the $PM_{10}$ concentration.

Figure 26 illustrates the over-sampling associated with the Method 201a stack sampling cyclone. In Figure 26, the performance characteristics of the $PM_{10}$ cyclone are defined by a $d_{50}$ of 11 μm and a slope of 1.76. The PSD characteristics are defined by a MMD of 20 μm and a GSD of 1.5. Based on these characteristics, the error relating to the mass of particles greater than 10 μm and deposited on the filter (mass 2) is 11.55 times the error relating to the mass of particles less than 10 μm in diameter that are captured by the pre-separator. In other words, the concentration measured by this sampler is 4.5 times the true $PM_{10}$ concentration.

Mathcad 2000 was also used to solve equation 25 for various PSD and sampler performance characteristics in order to obtain an initial concept of how the interaction of these characteristics impacts the concentration ratio. The PSD characteristics included in the evaluation were MMD's of 5 and 10 μm with a GSD of 1.5 and MMD's of 15 and 20 μm with a GSD of 2.0. The sampler performance characteristics included the nine combinations of $d_{50}$ and slope values for the ambient $PM_{10}$ sampler and the nine combinations of $d_{50}$ and slope values for the $PM_{2.5}$ sampler, as previously described. Table 19 lists the results of this evaluation for the ambient $PM_{10}$ and $PM_{2.5}$ sampler, respectively.

In addition to ratios of sampler to true concentrations, Table 19 contains estimates for sampler concentrations under the assumption that the current regulated limit is based on a sampler concentration and that the regulation should be based on a true concentration. In other words, the NAAQS are based on sampler concentrations; however, the NAAQS should be based on true concentrations so that all industries are equally regulated. The mathematical definition for this assumption is

$$C_{Acceptable} = Ratio * C_{NAAQS} \tag{30}$$

where $C_{NAAQS}$ corresponds to the current concentrations associated with the NAAQS and $C_{acceptable}$ corresponds to the acceptable concentrations *if* the NAAQS were based on true concentrations. The NAAQS for $PM_{10}$ and $PM_{2.5}$ are 150 and 65 μg/acm, respectively. The following conclusions can be drawn from Table 19: (1) the $PM_{10}$ sampler performance characteristics that define the range of acceptable concentrations are a $d_{50}$ of 9.5 μm with a slope of 1.4 and 1.6 and a $d_{50}$ of 10.5 μm with a slope of 1.4 and 1.6, (2) the $PM_{2.5}$ sampler performance characteristics that define the range of acceptable concentrations are a $d_{50}$ of 2.3 μm with a slope of 1.27 and a $d_{50}$ of 2.7 μm with a slope of 1.33 for PSD's characterized by MMDs greater than 2.5 μm, (3) the ratios for $PM_{10}$ range from 89 to 139%, (4) the ratios for $PM_{2.5}$ range from 108 to 1,314%, and (3) the ratio is equal to 100% only when the sampler $d_{50}$ is equal to the PSD's MMD. This initial evaluation was expanded to incorporate a larger range of MMDs and GSDs for the $PM_{10}$ and $PM_{2.5}$ ambient air samplers.

In order to define the differences in the simulated sampler measurements and true $PM_{10}$ concentrations, equations 18 and 20 were solved using Mathcad for a $d_{50}$ equal to 10.5 μm, slope of 1.6, GSD of 1.5, and MMDs ranging from 1 to 40 μm. Results of this simulation are illustrated in Figure 27. In Figure 27, three MMDs are highlighted. The first corresponds to a MMD of 5.7 μm, MMD associated with urban dust as defined by the EPA, and the other two correspond to the MMDs encompassing the range of MMDs expected from agricultural type dusts, MMDs of 15 and 25 μm. When comparing the sampled to true concentrations for the urban dust, the sampled concentration is about 9% [i.e. (true percent less than 5.7 μm – sampled percent less than 5.7 μm)/(true percent less than 5.7 μm)] lower than the true concentration. Further, when comparing the sampled to true concentrations for the range of agricultural type dusts, the sampled concentrations were 75 to 700 % higher than the true concentrations.

Figure 28 is similar to Figure 27, except that the GSD is set to a value of 2.0. When comparing the sampled to true concentrations for the urban dusts, the sampled concentration is about 3% lower than the true concentration. When comparing the sampled to true concentrations for the range of agricultural type dusts, the sampled concentrations were 20 to 61% higher than the true concentrations. Figure 29 compares $PM_{2.5}$ sampler measurements to true concentrations. The $PM_{2.5}$ sampler $d_{50}$ and slope were assumed to be 2.7 μm and 1.33, respectively. The GSD was set to a value of 2.0. When using these parameters, the sampled concentration for the urban dust was about 16% higher than the true concentration. For the agricultural type dusts, the sampled concentrations were 73 to 138% higher than the true concentration of $PM_{2.5}$.

To further describe how the interaction of the PSD and sampler performance characteristics affect the acceptable PM concentrations, a series of calculations were performed in Mathcad 2000 to generate a data file containing the solutions to equations 25 and 30 over a range of parameters. These PSD parameters included MMD values ranging from 1 to 40 μm (in increments of 1 μm), and GSD values ranging from 1.3 to 2.5 (in increments of 0.1). The sampler performance characteristics corresponded to the $PM_{10}$ ambient air sampler, $PM_{2.5}$ ambient air sampler with no $PM_{10}$ inlet, and the $PM_{2.5}$ ambient air sampler with a $PM_{10}$ inlet. The sampler performance characteristics also corresponded to the parameters defining the boundary tolerance ranges for the individual samplers. For example, $d_{50}$ values of 9.5 and 10.5 μm with slopes of 1.6 and 1.4 were used for the $PM_{10}$ ambient air sampler. To illustrate the results of this simulation, several graphs were created to demonstrate how each of the parameters affects the sampler to true concentration ratio.

**Table 19.** Differences between theoretical sampler and true concentrations for various particle size and sampler performance characteristics.

| | | Particle Size distribution (PSD) Characteristics | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | MMD = 5 µm<br>GSD = 1.5 | | MMD = 10 µm<br>GSD = 1.5 | | MMD = 15 µm<br>GSD = 2.0 | | MMD = 20 µm<br>GSD = 2.0 | |
| Cutpoint (µm) | Slope | Conc. (µg/m$^3$)[ζ] | Ratio[γ] | Conc. (µg/m$^3$)[ζ] | Ratio[γ] | Conc. (µg/m$^3$)[ζ] | Ratio[γ] | Conc. (µg/m$^3$)[ζ] | Ratio[γ] |
| *PM$_{10}$ sampler characteristics* | | | | | | | | | |
| 9.5 | 1.4 | 139.4 | 92.9% | **138.3** | 92.2% | **148.7** | 99.1% | **157.8** | 105.2% |
| 9.5 | 1.5 | 136.2 | 90.8% | 139.4 | 92.9% | 153.0 | 102.0% | 167.3 | 111.5% |
| 9.5 | 1.6 | **133.2** | 88.8% | 140.1 | 93.4% | 157.2 | 104.8% | 176.9 | 117.9% |
| 10.0 | 1.4 | 142.1 | 94.7% | 150.0 | 100.0% | 160.8 | 107.2% | 174.2 | 116.1% |
| 10.0 | 1.5 | 139.1 | 92.7% | 150.0 | 100.0% | 164.9 | 109.9% | 183.5 | 122.3% |
| 10.0 | 1.6 | 136.2 | 90.8% | 150.0 | 100.0% | 168.8 | 112.5% | 192.8 | 128.5% |
| 10.5 | 1.4 | **144.5** | 96.3% | **161.1** | 107.4% | 172.8 | 115.2% | 190.5 | 127.0% |
| 10.5 | 1.5 | 141.5 | 94.3% | 160.2 | 106.8% | 176.4 | 117.6% | 199.7 | 133.1% |
| 10.5 | 1.6 | 138.6 | 92.4% | 159.5 | 106.3% | **180.0** | 120.0% | **208.8** | 139.2% |
| *PM$_{2.5}$ sampler characteristics* | | | | | | | | | |
| 2.3 | 1.27 | **73.65** | 113.3% | **185.45** | 285.3% | **70.40** | 108.3% | **76.57** | 117.8% |
| 2.3 | 1.30 | 80.28 | 123.5% | 242.19 | 372.6% | 76.12 | 117.1% | 84.76 | 130.4% |
| 2.3 | 1.33 | 87.23 | 134.2% | 313.30 | 482.0% | 82.49 | 126.9% | 94.06 | 144.7% |
| 2.5 | 1.27 | 104.78 | 161.2% | 345.35 | 531.3% | 97.05 | 149.3% | 109.92 | 169.1% |
| 2.5 | 1.30 | 112.52 | 173.1% | 423.87 | 652.1% | 104.26 | 160.4% | 120.90 | 186.0% |
| 2.5 | 1.33 | 120.58 | 185.5% | 534.17 | 821.8% | 112.26 | 172.7% | 133.19 | 204.9% |
| 2.7 | 1.27 | 141.77 | 218.1% | 559.07 | 860.1% | 129.16 | 198.7% | 151.97 | 233.8% |
| 2.7 | 1.30 | 150.28 | 231.2% | 693.49 | 1066.9% | 138.06 | 212.4% | 166.01 | 255.4% |
| 2.7 | 1.33 | **159.12** | 244.8% | **854.10** | 1314.0% | **147.81** | 227.4% | **181.74** | 279.6% |

[ζ] *Values are based on the assumption that true concentrations are the correct estimates of the corresponding PM.*

[γ] *Concentrations are based on the corresponding regulations and adjusted by the ratio. Sampler concentrations for PM$_{10}$ and PM$_{2.5}$ are 150 and 65 µg/m$^3$, respectively.*

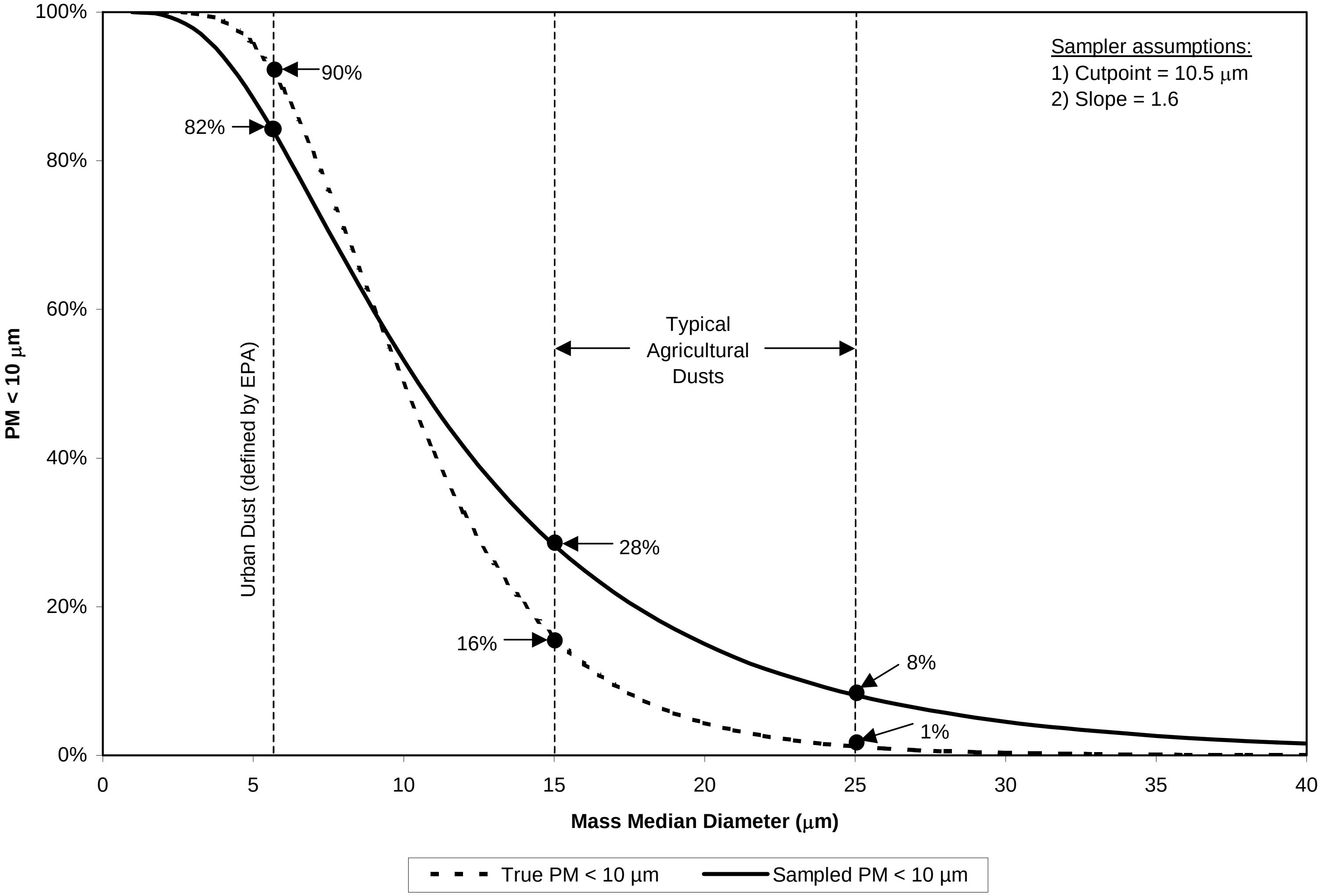


**Figure 27.** Comparison of true and sampled $PM_{10}$ percentages for a range of PSD mass median diameters and a GSD of 1.5.

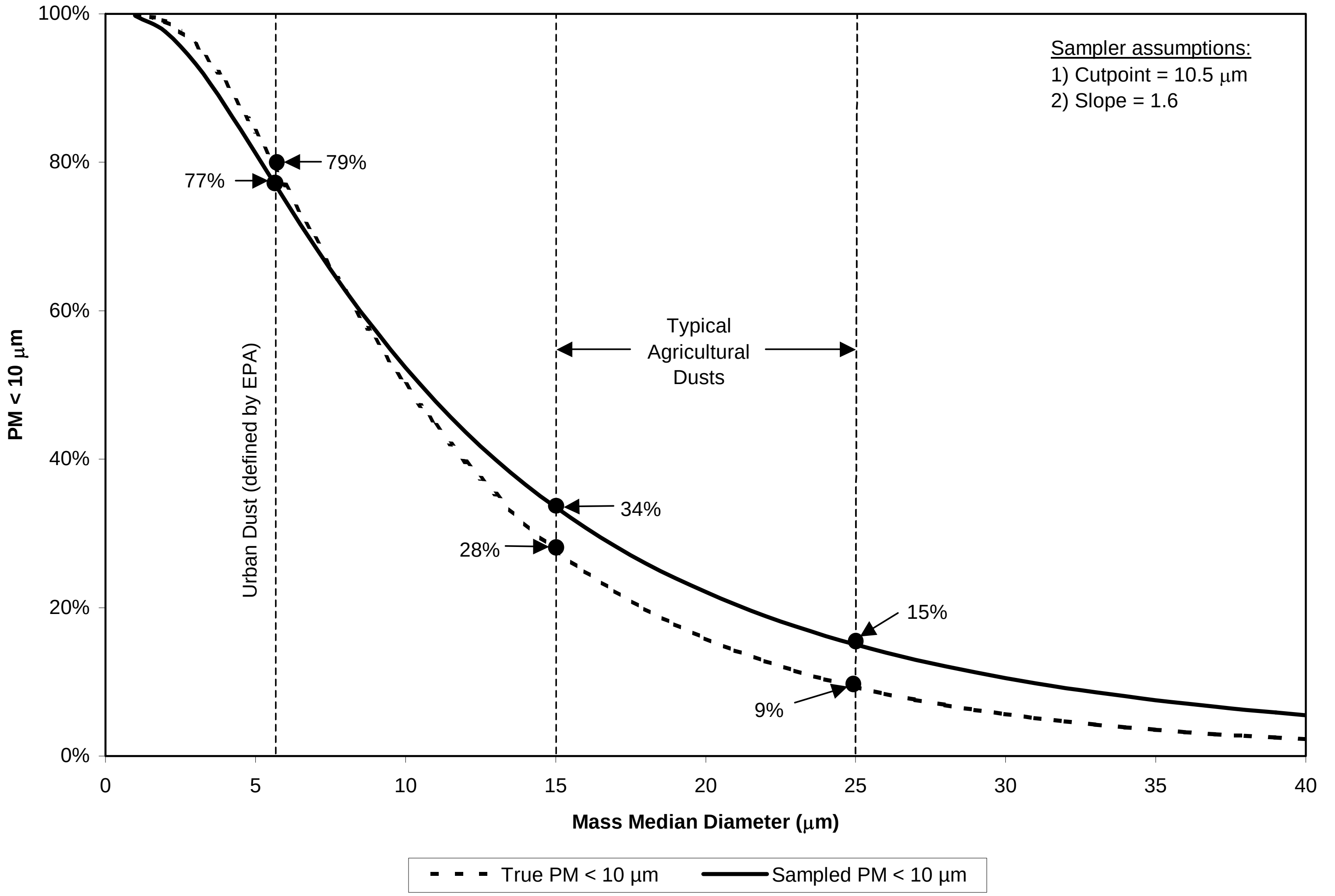


**Figure 28.** Comparison of true and sampled $PM_{10}$ percentages for a range of PSD mass median diameters and a GSD of 2.0.

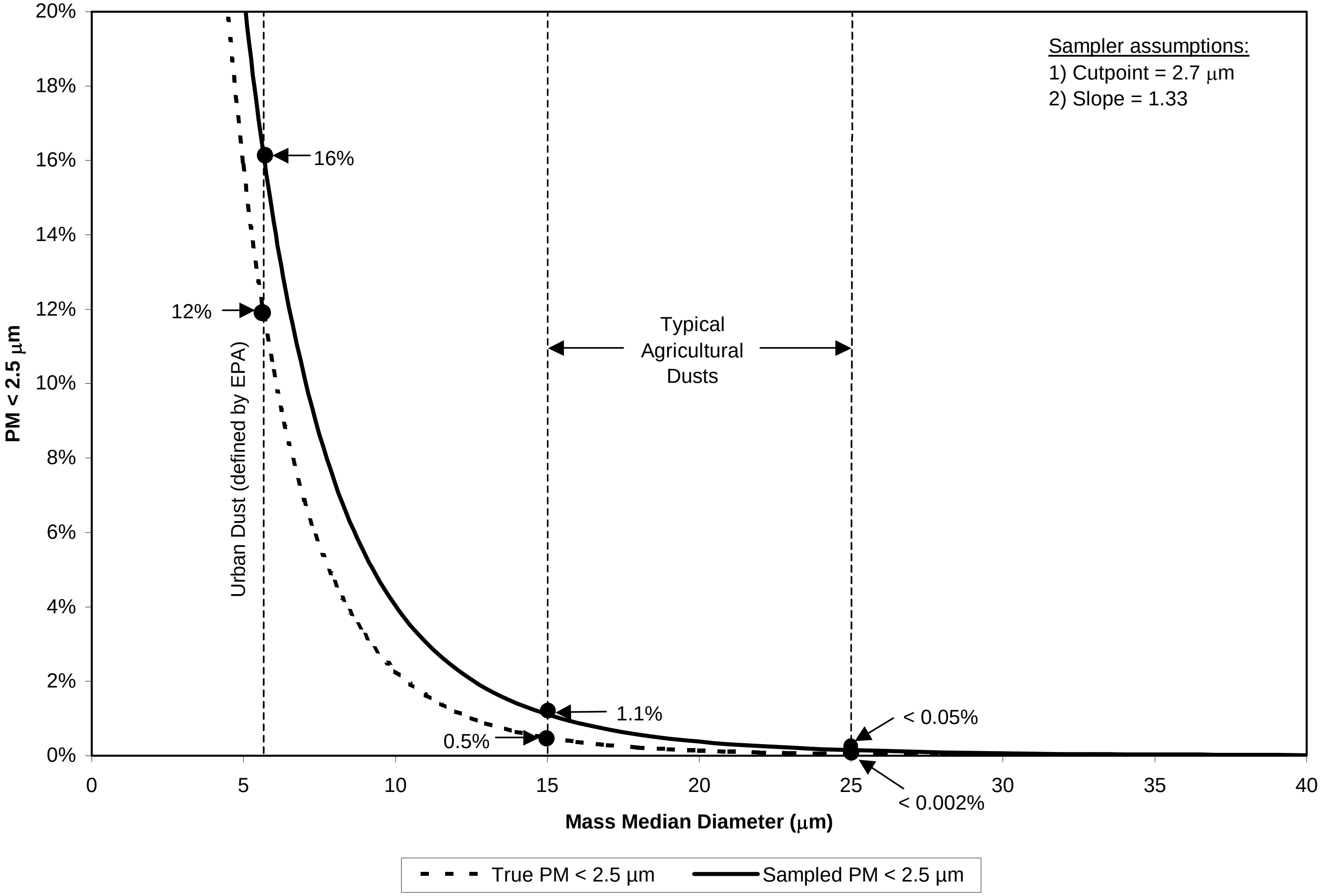
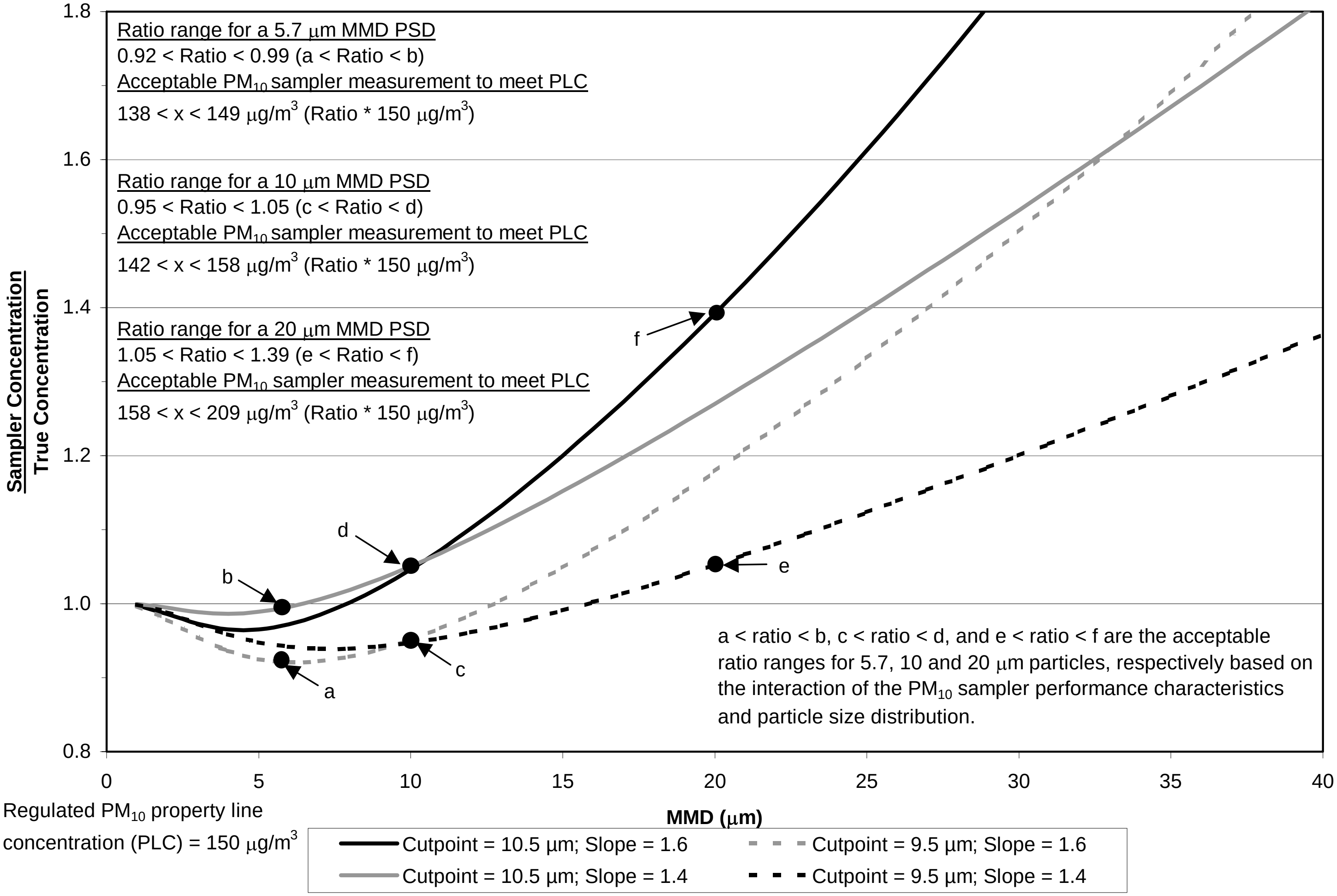


**Figure 29.** Comparison of true and sampled $PM_{2.5}$ percentages for a range of PSD mass median diameters and a GSD of 2.0.

In Figure 30, the GSD is held constant at 2.0 for the four sets of $PM_{10}$ sampler performance characteristics, which define the acceptable concentrations for $PM_{10}$, and the PSD MMDs range from 1 to 40 μm. To aid in the interpretation of the graph, an average concentration ratio is defined as the average of the largest and smallest ratio associated with the range of ratios defined by the sampler performance characteristics for a particular MMD. Conclusions that can be drawn from the information presented in Figure 30 are: 1) the average ratio is less than 1.0 when the MMD is less than the $d_{50}$, 2) the average ratio is equal to 1.0 when the MMD is equal to the $d_{50}$, 3) the average ratio is greater than 1.0 when the MMD is greater than the $d_{50}$, and 4) the ratio range increases as the MMD increases. In general terms, when the ratio is less than 1.0 the current method of regulating $PM_{10}$ under-estimates the concentration of PM less than or equal to 10 μm AED and when the ratio is greater than 1.0 the current method over-estimates the concentration of PM less than or equal to 10 μm AED. For example, if a PSD were characterized by a MMD of 5.7 μm AED and a GSD of 2.0, then the acceptable range of $PM_{10}$ sampler concentrations would be 138 to 149 $\mu g/m^3$ (i.e., ratios of 0.92 and 0.99 obtained from Figure 30 and multiplied by 150 $\mu g/m^3$ the current NAAQS for $PM_{10}$). In this scenario, the $PM_{10}$ sampler uncertainty is +/- 5.5 $\mu g/m^3$ and the sampler bias is 0 $\mu g/m^3$ if the sampler concentration is assumed to be the standard and -6.5 $\mu g/m^3$ if the true concentration is assumed to be the standard. Likewise, if the PSD were characterized by a MMD of 10 μm and a GSD of 2.0 then the acceptable range of $PM_{10}$ sampler concentrations would be 142 to 158 $\mu g/m^3$. This corresponds to a $PM_{10}$ sampler uncertainty of +/- 8.0 $\mu g/m^3$ and a sampler bias of 0 $\mu g/m^3$ if the sampler concentration is assumed to be the standard and 0 $\mu g/m^3$ if the true concentration is assumed to be the standard. Further, if the PSD were characterized by a MMD of 20 μm and a GSD of 2.0, then the acceptable $PM_{10}$ sampler concentrations would be 158 to 209 $\mu g/m^3$. Corresponding to a $PM_{10}$ sampler uncertainty of +/- 25.5 $\mu g/m^3$ and a sampler bias is 0 $\mu g/m^3$ if the sampler concentration is assumed to be the standard and 33.5 $\mu g/m^3$ if the true concentration is assumed to be the standard.

The data presented in Figure 31 are based on the same assumptions as Figure 30, except the data are based on a GSD of 1.5. When comparing Figures 30 and 31, it is obvious that the ratios increase much more rapidly as the MMD increases when the GSD is 1.5 as compared to a GSD of 2.0. For example, if a PSD were characterized by a MMD of 5.7 μm and a GSD of 1.5, then the acceptable range of $PM_{10}$ sampler concentrations would be 131 to 144 $\mu g/m^3$. In this scenario, the $PM_{10}$ sampler uncertainty is +/- 6.5 $\mu g/m^3$ and the sampler bias is 0 $\mu g/m^3$ if the sampler concentration is assumed to be the standard and -12.5 $\mu g/m^3$ if the true concentration is assumed to be the standard. If a PSD were characterized by a MMD of 10 μm and a GSD of 1.5, then the acceptable range of $PM_{10}$ sampler concentrations would be 138 to 161 $\mu g/m^3$. This corresponds to a $PM_{10}$ sampler uncertainty of +/- 11.5 $\mu g/m^3$ and a sampler bias of 0 $\mu g/m^3$ if the sampler concentration is assumed to be the standard and 0 $\mu g/m^3$ if the true concentration is assumed to be the standard. Further, if the PSD were characterized by a MMD of 20 μm and a GSD of 1.5 then the acceptable range of $PM_{10}$ sampler concentrations would be 271 to 514 $\mu g/m^3$ (i.e., ratios of 1.81 and 3.43 obtained from Figure 31 and multiplied by 150 $\mu g/m^3$ the current NAAQS for $PM_{10}$). Corresponding to a $PM_{10}$ sampler uncertainty of +/- 121.5 $\mu g/m^3$ and a sampler bias is 0 $\mu g/m^3$ if the sampler concentration is assumed to be the standard and 242.5 $\mu g/m^3$ if the true concentration is assumed to be the standard. Another conclusion that can be drawn from the data presented in Figures 30 and 31 is that the range of acceptable concentrations increases as the GSD increases.

Figure 32 is a generalized graph to illustrate how MMD's and GSD's affect the concentration ratios for a $PM_{10}$ sampler with a $d_{50}$ of 10.0 μm and a slope of 1.5. The general observation that should be made from this graph is that the concentration ratios decrease (ratio approaches 1.0) as the GSD increases. Figure 33 further expands on how the concentration ratios are impacted by GSD.



Ratio range for a 5.7 µm MMD PSD
0.92 < Ratio < 0.99 (a < Ratio < b)
Acceptable $PM_{10}$ sampler measurement to meet PLC
138 < x < 149 $\mu g/m^3$ (Ratio * 150 $\mu g/m^3$)

Ratio range for a 10 µm MMD PSD
0.95 < Ratio < 1.05 (c < Ratio < d)
Acceptable $PM_{10}$ sampler measurement to meet PLC
142 < x < 158 $\mu g/m^3$ (Ratio * 150 $\mu g/m^3$)

Ratio range for a 20 µm MMD PSD
1.05 < Ratio < 1.39 (e < Ratio < f)
Acceptable $PM_{10}$ sampler measurement to meet PLC
158 < x < 209 $\mu g/m^3$ (Ratio * 150 $\mu g/m^3$)

a < ratio < b, c < ratio < d, and e < ratio < f are the acceptable ratio ranges for 5.7, 10 and 20 µm particles, respectively based on the interaction of the $PM_{10}$ sampler performance characteristics and particle size distribution.

Regulated $PM_{10}$ property line concentration (PLC) = 150 $\mu g/m^3$

**Figure 30.** Theoretical ratios of $PM_{10}$ sampler to true PSD concentrations (PSD – GSD = 2.0).

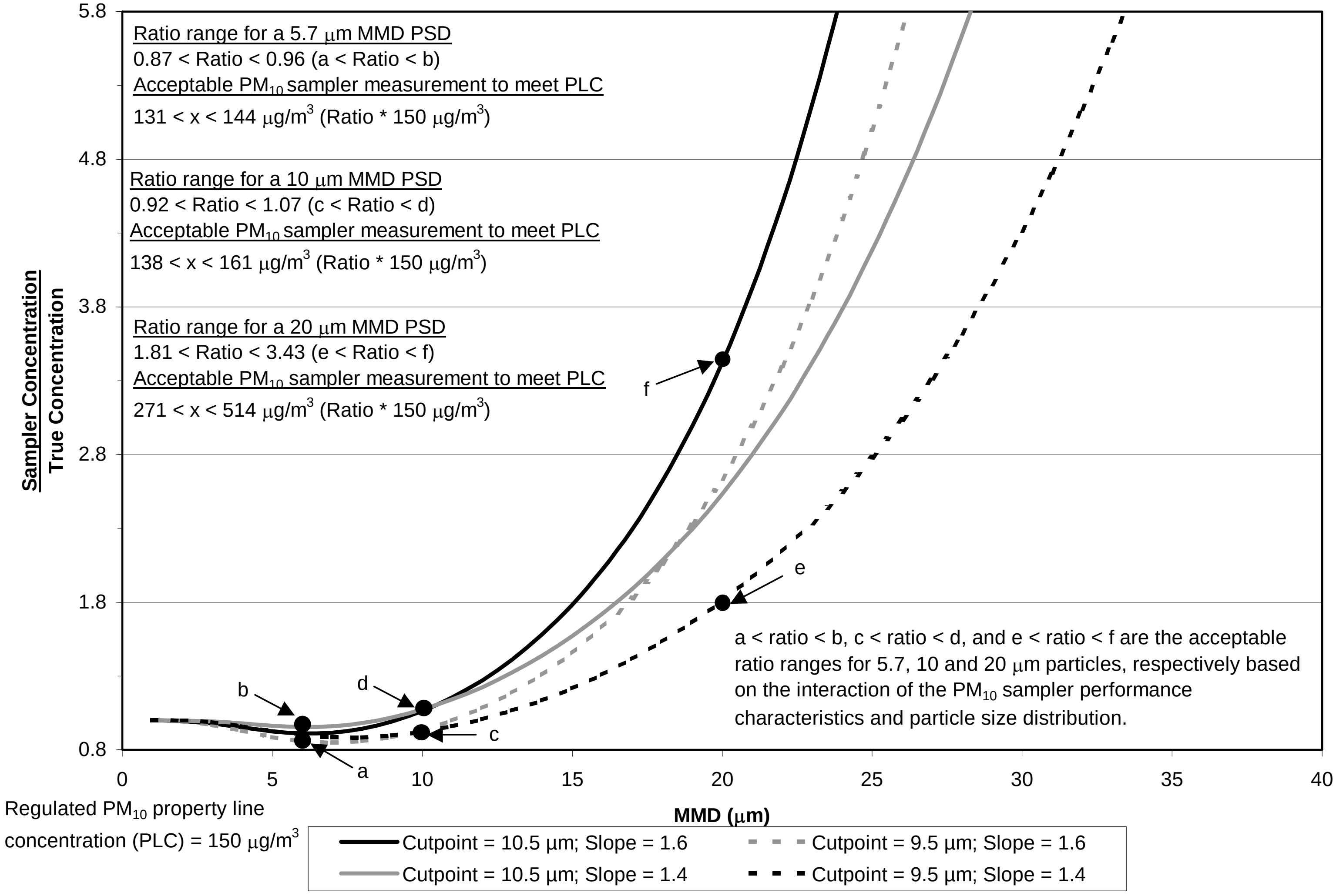


**Figure 31.** Theoretical ratios of PM$_{10}$ sampler to true PSD concentrations (PSD – GSD = 1.5).

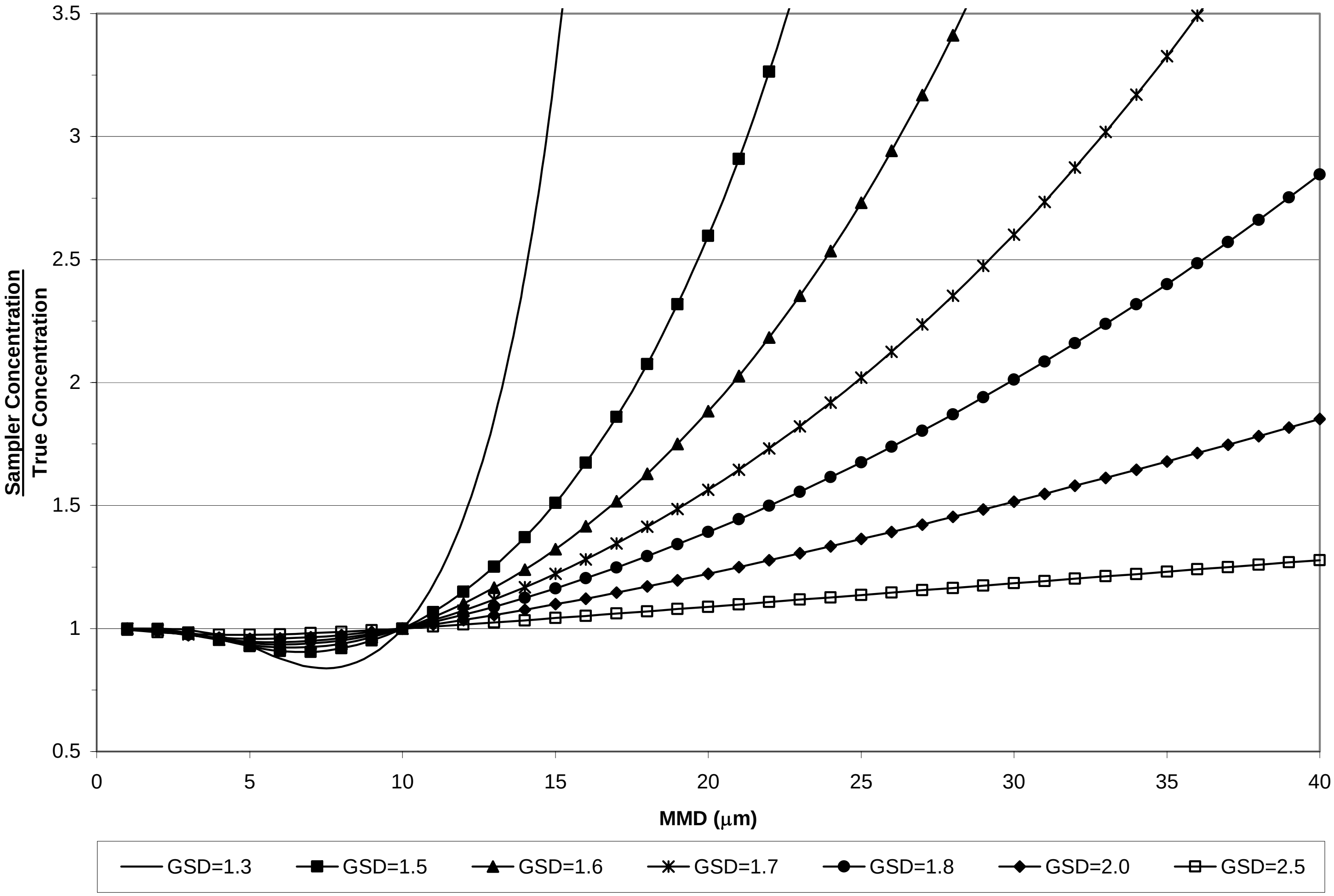
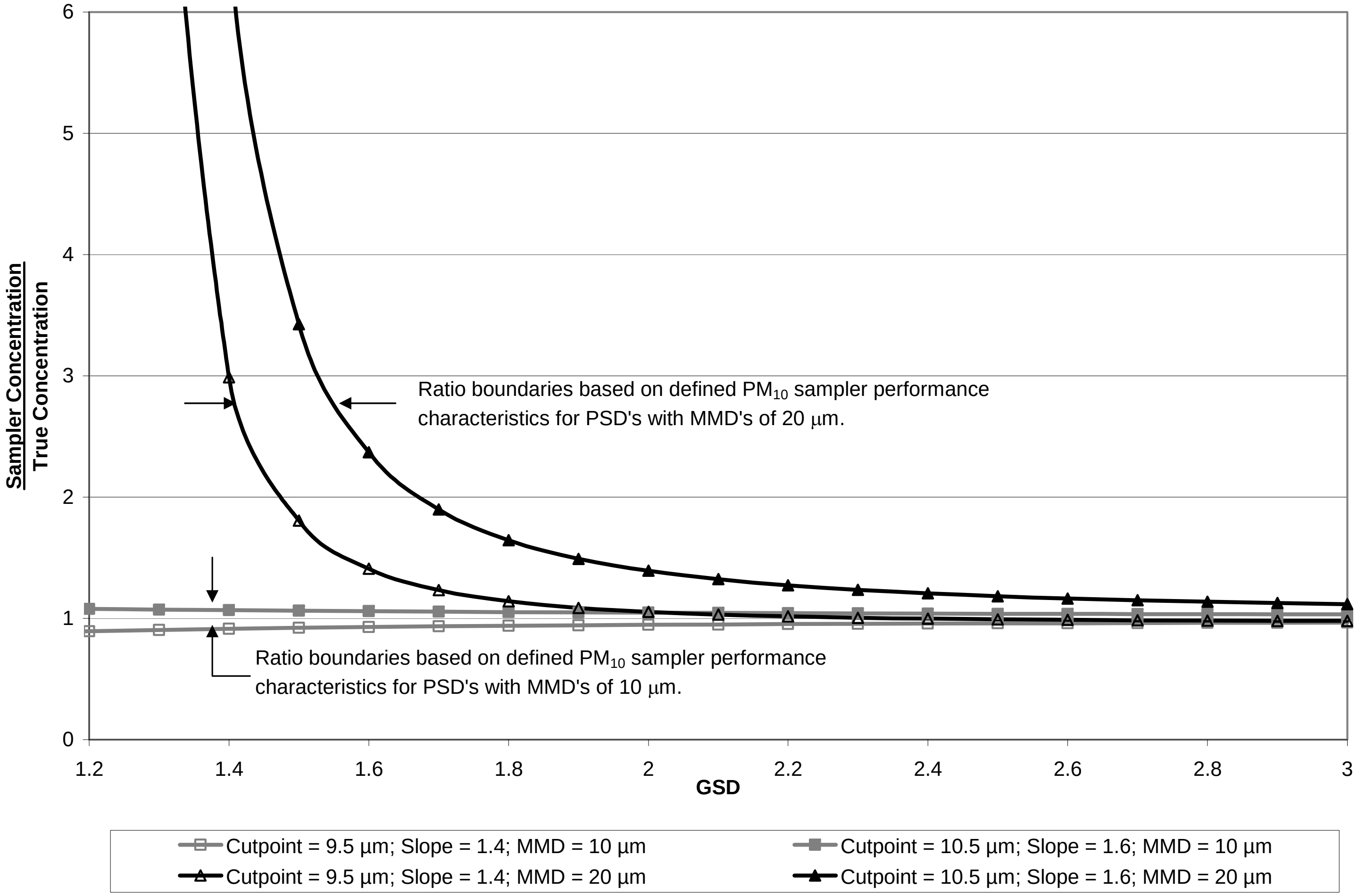


**Figure 32.** Theoretical ratios of $PM_{10}$ sampler to true PSD concentrations ($PM_{10}$ sampler characteristics; cutpoint = 10 μm and slope = 1.5).



Ratio boundaries based on defined PM$_{10}$ sampler performance characteristics for PSD's with MMD's of 20 μm.

Ratio boundaries based on defined PM$_{10}$ sampler performance characteristics for PSD's with MMD's of 10 μm.

**Figure 33.** Theoretical PM$_{10}$ sampler to true concentration ratio boundaries based on varying GSDs for PSDs with MMDs of 10 and 20 μm.

The data presented in Figure 33 are based on MMD's of 10 and 20 μm, sampler performance characteristics of $d_{50}$ equal to 9.5 μm with a slope of 1.4 and $d_{50}$ equal to 10.5 μm with a slope of 1.6, and variable GSD's ranging from 1.2 to 3.0. The general conclusions that should be drawn from Figure 33 include: 1) when the MMD is equal to the $d_{50}$, the range of concentration ratios is centered around 1.0 for all GSDs, 2) as the GSD increases, the concentration ratio decreases and approaches 1.0, and 3) as the GSD decreases, the concentration ratio increases and approaches infinity for an MMD of 20 μm AED.

Similar to the $PM_{10}$ sampler simulations, the interaction of the PSD and $PM_{2.5}$ sampler (with and without a $PM_{10}$ inlet) performance characteristics were simulated through a series of calculations performed in Mathcad 2000 to generate a data file containing the solutions to equations 25 and 30 over a range of parameters. These parameters included MMD values ranging from 1 to 40 μm (in increments of 1 μm) and GSD values ranging from 1.3 to 2.5 (in increments of 0.1). To illustrate the results of this simulation, several graphs were created to demonstrate how each of the parameters affects the sampler to true concentration ratio.

In Figure 34, the GSD is held constant at 2.0 for the two sets of $PM_{2.5}$ sampler (no $PM_{10}$ inlet) performance characteristics that define the acceptable concentrations for $PM_{2.5}$, and the PSD MMDs ranged from 1 to 40 μm. To aid in the interpretation of the graph, an average concentration ratio is defined as the average of the largest and smallest ratio associated with the range of ratios defined by the sampler performance characteristics for a particular MMD. Conclusion that can be drawn from the information presented in Figure 34 are: 1) the average ratio is equal to 1.0 when the MMD is equal to the $d_{50}$, 2) the average ratio is greater than 1.0 with the MMD is greater than the $d_{50}$, and 3) the ratio range increases as the MMD increases. In general terms, when the ratio is equal to 1.0, the current method of regulating $PM_{2.5}$ results in an exact concentration measurement of PM less than or equal to 2.5 μm AED and when the ratio is greater than 1.0, the current method over-estimates the concentration of PM less than 2.5 μm AED. For example, if a PSD were characterized by a MMD of 5.7 μm and a

GSD of 2.0, then the acceptable range of $PM_{2.5}$ sampler concentrations would be 60 to 87 μg/$m^3$ (i.e. ratios of 0.92 and 1.34 obtained from Figure 34 and multiplied by 65 μg/$m^3$, the proposed NAAQS for $PM_{2.5}$). In this scenario, the $PM_{2.5}$ sampler uncertainty is +/- 13.5 μg/$m^3$ and since EPA essentially states that the $PM_{2.5}$ should correspond to a true concentration, the $PM_{2.5}$ sampler bias is 8.5 μg/$m^3$. If the PSD were characterized by a MMD of 10 μm and a GSD of 2.0, then the acceptable range of $PM_{2.5}$ sampler concentrations would be 64 to 115 μg/$m^3$. The corresponding $PM_{2.5}$ sampler uncertainty of +/- 25.5 μg/$m^3$ and a bias of 24.5 μg/$m^3$. Further, if the PSD were characterized by a MMD of 20 μm and a GSD of 2.0, then the acceptable range of $PM_{2.5}$ sampler concentrations would be 77 to 182 μg/$m^3$. Corresponding to a $PM_{2.5}$ sampler uncertainty of +/- 52.5 μg/$m^3$ and a bias of 64.5 μg/$m^3$.

The data presented in Figure 35 are based on the same assumptions as Figure 34, except the data are based on a GSD of 1.5. When comparing Figures 34 and 35, it is obvious that the ratios increase much more rapidly as the MMD increases when the GSD is 1.5 as compared to a GSD of 2.0. For example, if a PSD were characterized by a MMD of 5.7 μm AED and a GSD of 1.5 then the acceptable range of $PM_{2.5}$ sampler concentrations would be 81 to 193 μg/$m^3$ (i.e. ratios of 1.24 and 2.96 obtained from Figure 35 and multiplied by 65 μg/$m^3$, the proposed NAAQS for $PM_{2.5}$). Corresponding to a $PM_{2.5}$ sampler uncertainty of +/- 56 μg/$m^3$ and a bias of 72 μg/$m^3$. If the PSD were characterized by a MMD of 10 μm AED and a GSD of 1.5, then the acceptable range of $PM_{2.5}$ sampler concentrations would be 185 to 854 μg/$m^3$. Corresponding to a $PM_{2.5}$ sampler uncertainty of +/- 334.5 μg/$m^3$ and a bias of 454.5 μg/$m^3$. Further, if the PSD were characterized by a MMD of 20 μm AED and a GSD of 1.5, then the acceptable range of $PM_{2.5}$ sampler concentrations would be 963 to 11,929 μg/$m^3$ (i.e. ratios of 14.81 and 193.5 obtained from Figure 35 and multiplied by 65 μg/$m^3$, the proposed NAAQS for $PM_{2.5}$). Corresponding to a $PM_{2.5}$ sampler uncertainty of +/- 5,483 μg/$m^3$ and a bias of 6,381 μg/$m^3$. Another conclusion drawn from the data presented in Figures 34 and 35 is that the range of acceptable concentrations increases as the GSD increases.

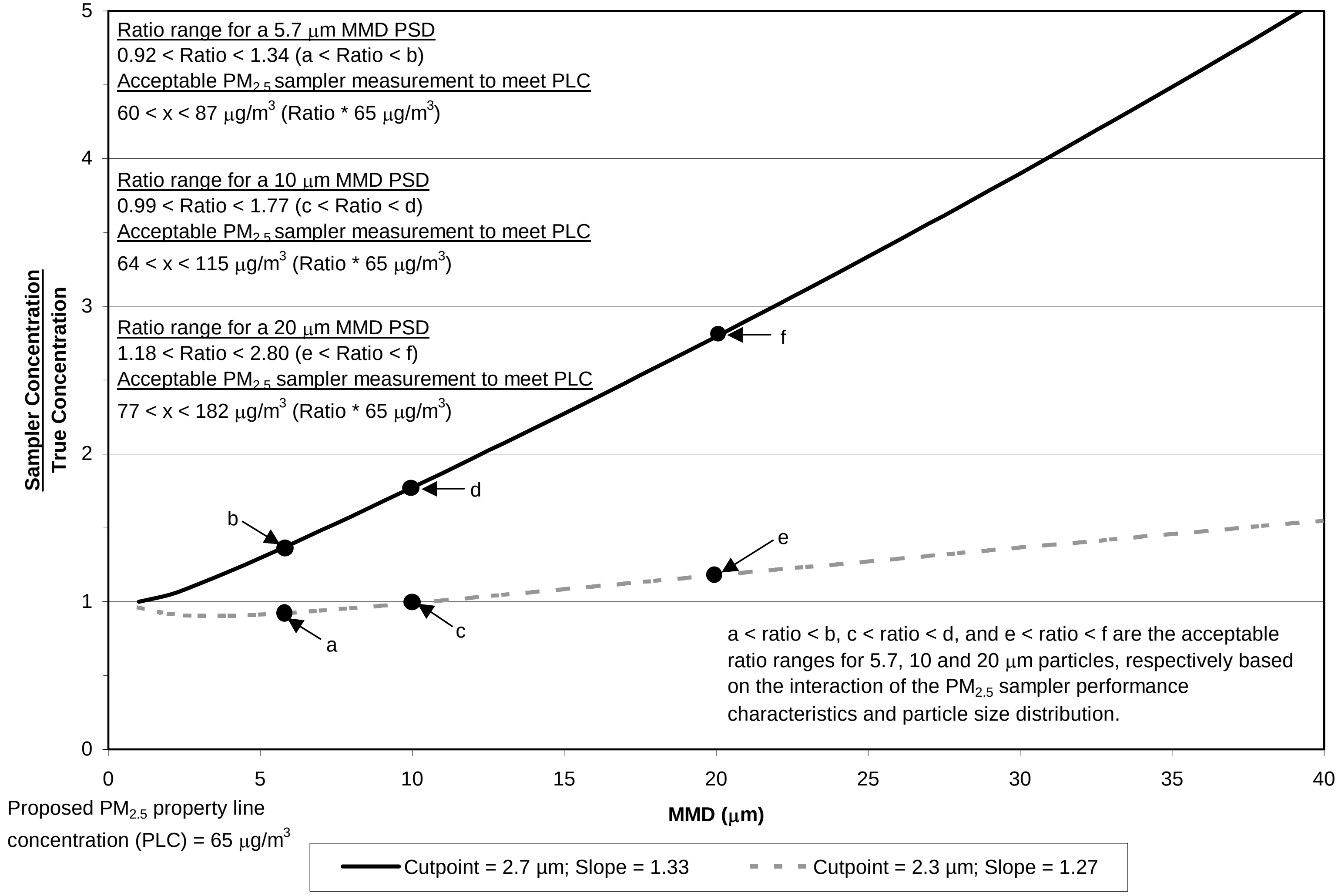


**Figure 34.** Theoretical ratios of $PM_{2.5}$ sampler to true PSD concentrations (PSD – GSD = 2.0).

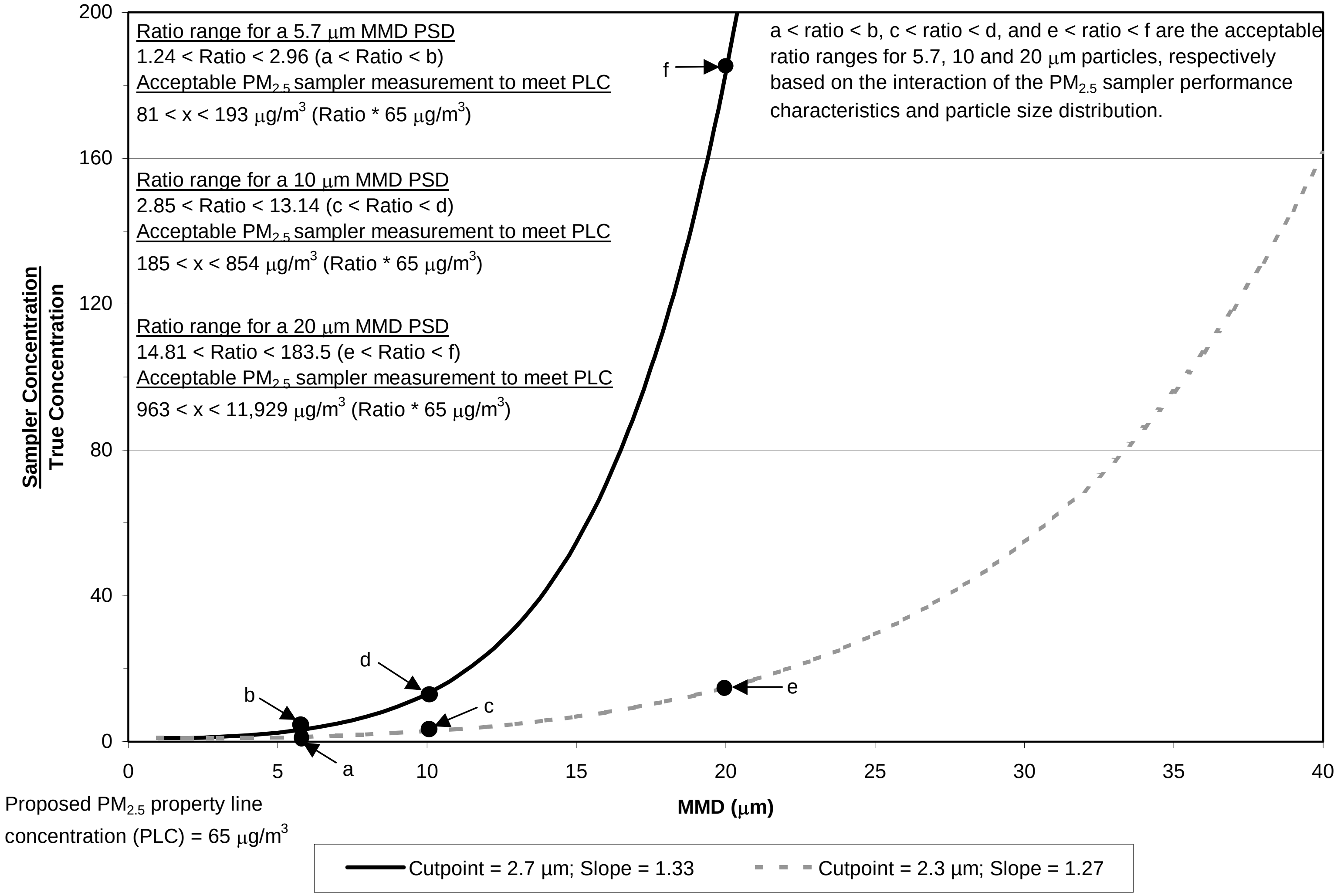
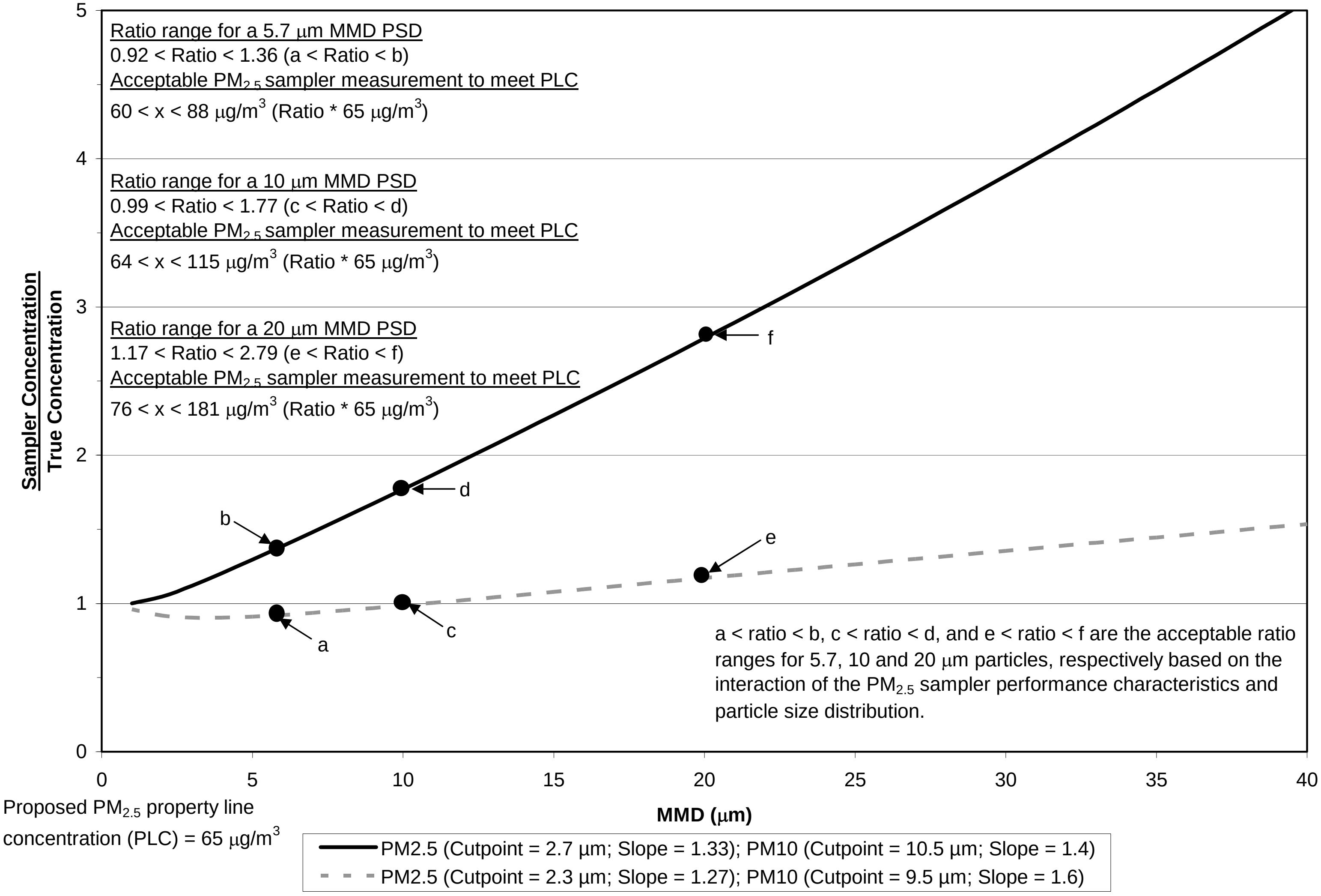


**Figure 35.** Theoretical ratios of $PM_{2.5}$ sampler to true PSD concentrations (PSD – GSD = 1.5).

The data presented in Figure 36 are based on the same assumptions as Figure 34, expect the sampler performance characteristics of a $PM_{10}$ inlet are incorporated into the simulation. The $d_{50}$ and slope values for the $PM_{10}$ inlet used in the simulation were determined through a trial and error process (i.e. determining which $PM_{10}$ $d_{50}$ and slope values coupled with the $PM_{2.5}$ boundary performance characteristic generated the $PM_{2.5}$ sampler with a $PM_{10}$ inlet boundary performance characteristics). The resulting $PM_{10}$ performance characteristics were defined as a $d_{50}$ of 9.5 μm with a slope of 1.6 and a $d_{50}$ of 10.5 μm with a slope of 1.4.

In general, the inclusion of the $PM_{10}$ inlet on the $PM_{2.5}$ sampler had very little effect on the sampler to true concentration ratio. For example, if the PSD were characterized by a MMD of 5.7 μm AED and a GSD of 2.0, then the acceptable concentration range for a $PM_{2.5}$ sampler with a $PM_{10}$ inlet would be 60 to 88 $\mu g/m^3$, as compared to 60 to 87 $\mu g/m^3$ for a $PM_{2.5}$ sampler with no $PM_{10}$ inlet. If the PSD were characterized by a MMD of 20 μm AED and a GSD of 2.0, then the acceptable concentration range for a $PM_{2.5}$ sampler with a $PM_{10}$ inlet would be 76 to 181 $\mu g/m^3$, as compared to 77 to 182 $\mu g/m^3$ for a $PM_{2.5}$ sampler with no $PM_{10}$ inlet.

The data presented in Figure 37 are based on the same assumptions as Figure 36, except the data are based on a GSD of 1.5. The inclusion of the $PM_{10}$ inlet on the $PM_{2.5}$ sampler had very little relative effect on the sampler to true concentration ratio. For example, if the PSD were characterized by a MMD of 5.7 μm AED and a GSD of 1.5, then the acceptable concentration range for a $PM_{2.5}$ sampler with a $PM_{10}$ inlet would be 83 to 204 $\mu g/m^3$, as compared to 81 to 193 $\mu g/m^3$ for a $PM_{2.5}$ sampler with no $PM_{10}$ inlet. If the PSD were characterized by a MMD of 20 μm AED and a GSD of 1.5 then the acceptable concentration range for a $PM_{2.5}$ sampler with a $PM_{10}$ inlet would be 928 to 11,557 $\mu g/m^3$, as compared to 963 to 11,929 $\mu g/m^3$ for a $PM_{2.5}$ sampler with no $PM_{10}$ inlet.



Ratio range for a 5.7 μm MMD PSD
0.92 < Ratio < 1.36 (a < Ratio < b)
Acceptable $PM_{2.5}$ sampler measurement to meet PLC
60 < x < 88 μg/$m^3$ (Ratio * 65 μg/$m^3$)

Ratio range for a 10 μm MMD PSD
0.99 < Ratio < 1.77 (c < Ratio < d)
Acceptable $PM_{2.5}$ sampler measurement to meet PLC
64 < x < 115 μg/$m^3$ (Ratio * 65 μg/$m^3$)

Ratio range for a 20 μm MMD PSD
1.17 < Ratio < 2.79 (e < Ratio < f)
Acceptable $PM_{2.5}$ sampler measurement to meet PLC
76 < x < 181 μg/$m^3$ (Ratio * 65 μg/$m^3$)

a < ratio < b, c < ratio < d, and e < ratio < f are the acceptable ratio ranges for 5.7, 10 and 20 μm particles, respectively based on the interaction of the $PM_{2.5}$ sampler performance characteristics and particle size distribution.

Proposed $PM_{2.5}$ property line concentration (PLC) = 65 μg/$m^3$

**Figure 36.** Theoretical ratios of $PM_{2.5}$ sampler, with $PM_{10}$ inlet, to true PSD concentrations (PSD – GSD = 2.0).

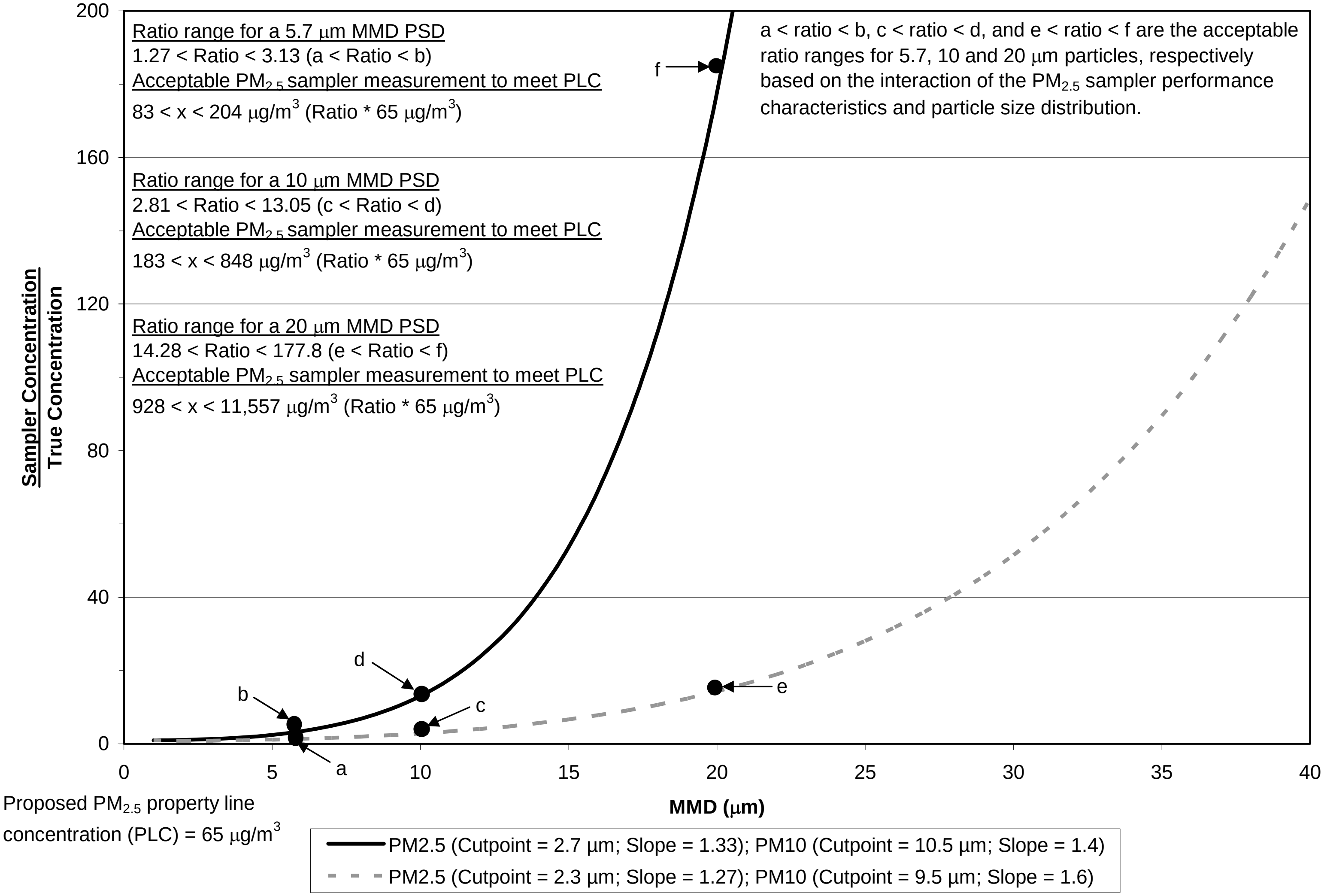


**Figure 37.** Theoretical ratios of $PM_{2.5}$ sampler, with $PM_{10}$ inlet, to true PSD concentrations (PSD – GSD = 1.5).

Although a decrease of 372 μg/m$^3$, when comparing the PM$_{2.5}$ sampler with and without a PM$_{10}$ inlet for a PSD characterized by a MMD of 20 μm and a GSD of 1.5, would seem significant, the relative difference is negligible (i.e. only about a 3% decrease). Therefore, only the PM$_{2.5}$ sampler performance characteristics, without the inclusion of the PM$_{10}$ inlet performance characteristics, were used in the remainder of the inherent sampler errors discussion.

Figure 38 is a generalized graph used to illustrate how MMD's and GSD's affect the concentration ratios for a PM$_{2.5}$ sampler with a d$_{50}$ of 2.5 μm and a slope of 1.3. The general observation that should be made from this graph is that the concentration ratios decrease (ratio approaches 1.0) as the GSD increases. Figure 39 further expands on how the concentration ratios are impacted by GSD. The data presented in Figure 39 are based on MMDs of 10 and 20 μm, sampler performance characteristic of a d$_{50}$ equal to 2.3 μm with a slope of 1.27 and a d$_{50}$ of 2.7 μm with a slope of 1.33, and variable GSD's ranging from 1.2 to 3.0. The general conclusions that should be drawn from this graph include: 1) as the GSD increases, the concentration ratio decreases and approaches 1.0 and 2) as the GSD decreases, the concentration ratio increases and approaches infinity.

According to the literature, an appropriate method of determining PM$_{coarse}$ is to subtract PM$_{2.5}$ sampler concentrations from PM$_{10}$ sampler concentrations. A data set was generated using PM$_{10}$ sampler performance characteristics of a d$_{50}$ equal to 9.5 μm with a slope of 1.4 and a d$_{50}$ equal to 10.5 μm with a slope of 1.6 along with PM$_{2.5}$ sampler performance characteristics of a d$_{50}$ of 2.7 μm with a slope of 1.33 and a d$_{50}$ of 2.3 μm with a slope of 1.27 in order to estimate the inherent PM$_{coarse}$ sampler errors attributed to the interaction of particle size and sampler performance characteristics. Ratios of sampler and true concentrations were calculated for PM$_{coarse}$ using the subtraction method for a range of PSD characteristics. The PSDs were characterized by MMDs ranging from 1 to 40 μm and GSDs of 1.5 and 2.0. The ratios determined from the series of calculations were multiplied by 85 μg/m$^3$ in order to estimate an acceptable PM$_{coarse}$ concentration. Note: PM$_{coarse}$ is not currently listed in the NAAQS and 85 μg/m$^3$ does not correspond to any PM$_{coarse}$ standard; 85 μg/m$^3$ was determined by subtracting the proposed $PM_{2.5}$ NAAQS from the current $PM_{10}$ NAAQS. Results of the simulation for a constant GSD of 2.0 and 1.5 are illustrated in Figures 40 and 41, respectively.

The data presented in Figure 40 corresponds to the ratio of sampled $PM_{coarse}$ to true $PM_{coarse}$ concentrations as a function of MMD with a constant GSD of 2.0. When comparing the data presented in Figure 30 (ratios for a $PM_{10}$ sampler with a constant GSD of 2.0) to the data presented in Figure 40, the ratio patterns appear to be similar. For example, if the PSD were characterized by a MMD of 5.7 μm AED and a GSD of 2.0, then the ratio of sampler to true concentration for $PM_{coarse}$ would be 0.85 to 1.0, as compared to 0.92 to 0.99 for the $PM_{10}$ sampler. If the PSD were characterized by a MMD of 20 μm and a GSD of 2.0, then the ratio of sampler to true concentration for $PM_{coarse}$ would be 1.04 to 1.39, compared to 1.05 to 1.39 for the $PM_{10}$ sampler. Conclusions that can be drawn from Figure 40 are that as the MMD of the dust being sampled increases, the ratio of sampled to true $PM_{coarse}$ more closely follows the trends associated with the ratio of sampler to true $PM_{10}$ concentrations. This conclusion is logical since the percent of $PM_{2.5}$, for a PSD characterized by a GSD of 2.0, decreases as the MMD increases. In other words, the errors associated with the $PM_{10}$ sampler are dominating the errors associated with the $PM_{2.5}$ sampler.

The data presented in Figure 41 is similar to the data presented in Figure 40, except the PSD is characterized by a GSD of 1.5. When comparing the data presented in Figure 31 (ratios for a $PM_{10}$ sampler with a constant GSD of 1.5) to the data presented in Figure 41, the ratio patterns are very similar. For example, if the PSD were characterized by a MMD of 5.7 μm AED and a GSD of 1.5, then the ratio of sampler to true concentration for $PM_{coarse}$ would be 0.81 to 0.94, as compared to 0.87 to 0.96 for the $PM_{10}$ sampler. If the PSD were characterized by a MMD of 10 or 20 μm with a GSD of 1.5 then the ratio of sampler to true concentration for $PM_{coarse}$ would be exactly the same as that for the $PM_{10}$ sampler. Conclusions that can be drawn from comparing Figure 30, 31, 40, and 41 are that as the GSD of the dust being sampled decreases, the more impact the $PM_{10}$ sampler errors have on the $PM_{coarse}$ concentrations.

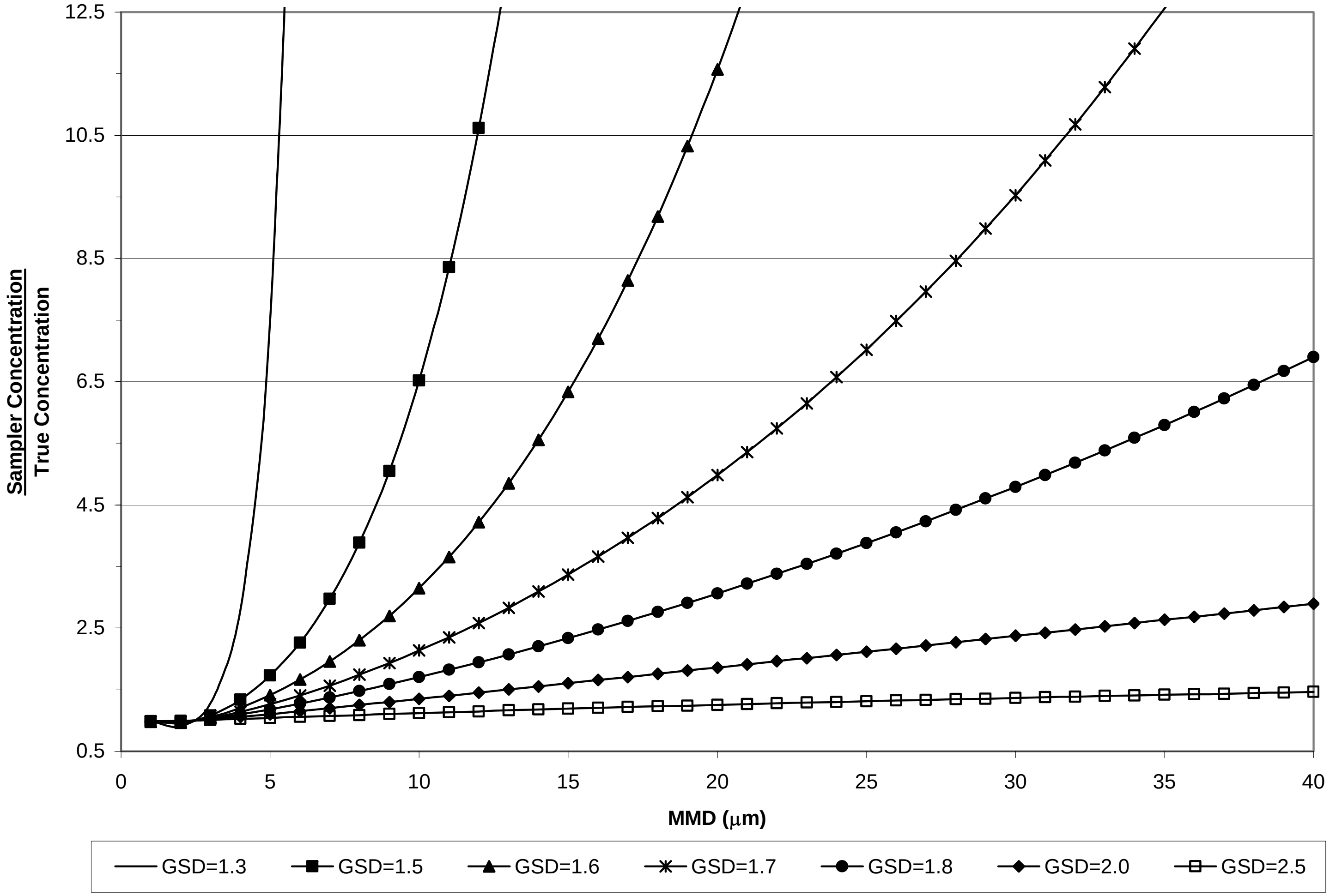


**Figure 38.** Theoretical ratios of $PM_{2.5}$ sampler to true PSD concentrations ($PM_{2.5}$ sampler characteristics; cutpoint = 2.5 μm and slope = 1.3).

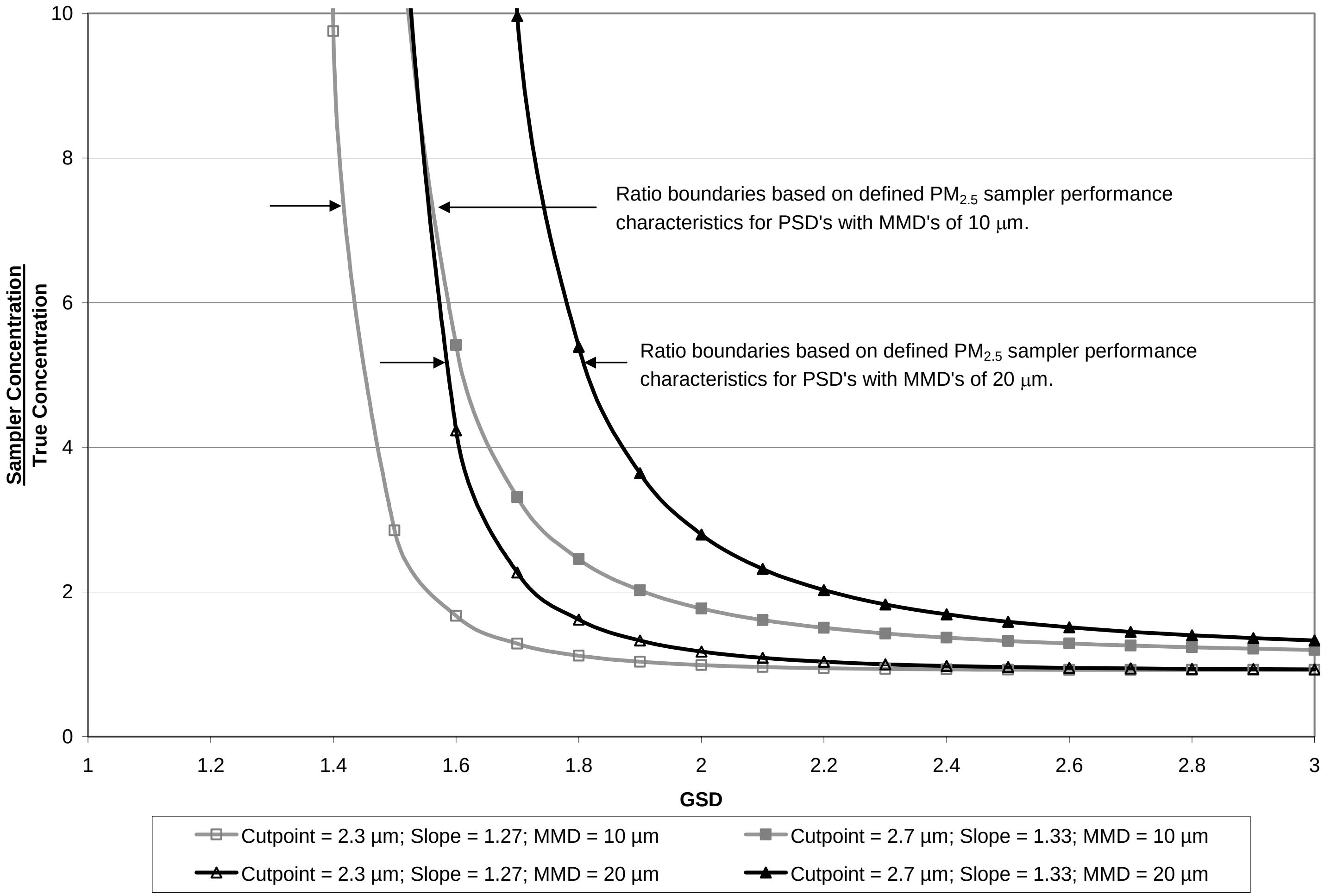


**Figure 39.** Theoretical $PM_{2.5}$ sampler to true concentration ratio boundaries based on varying GSDs for PSDs with MMDs of 10 and 20 µm.

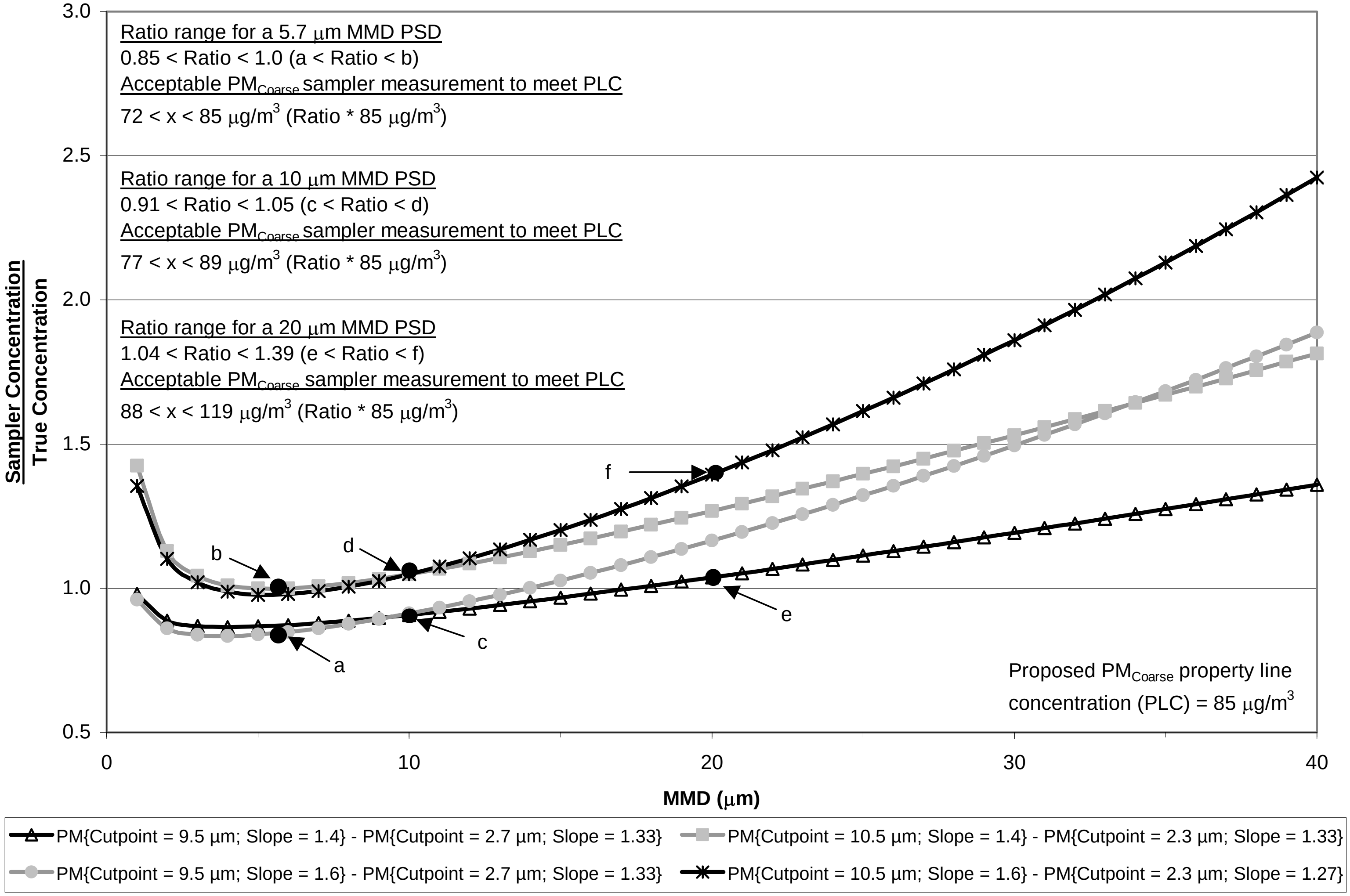


**Figure 40.** Theoretical ratios of $PM_{Coarse}$ to true PSD concentrations (PSD – GSD = 2.0).

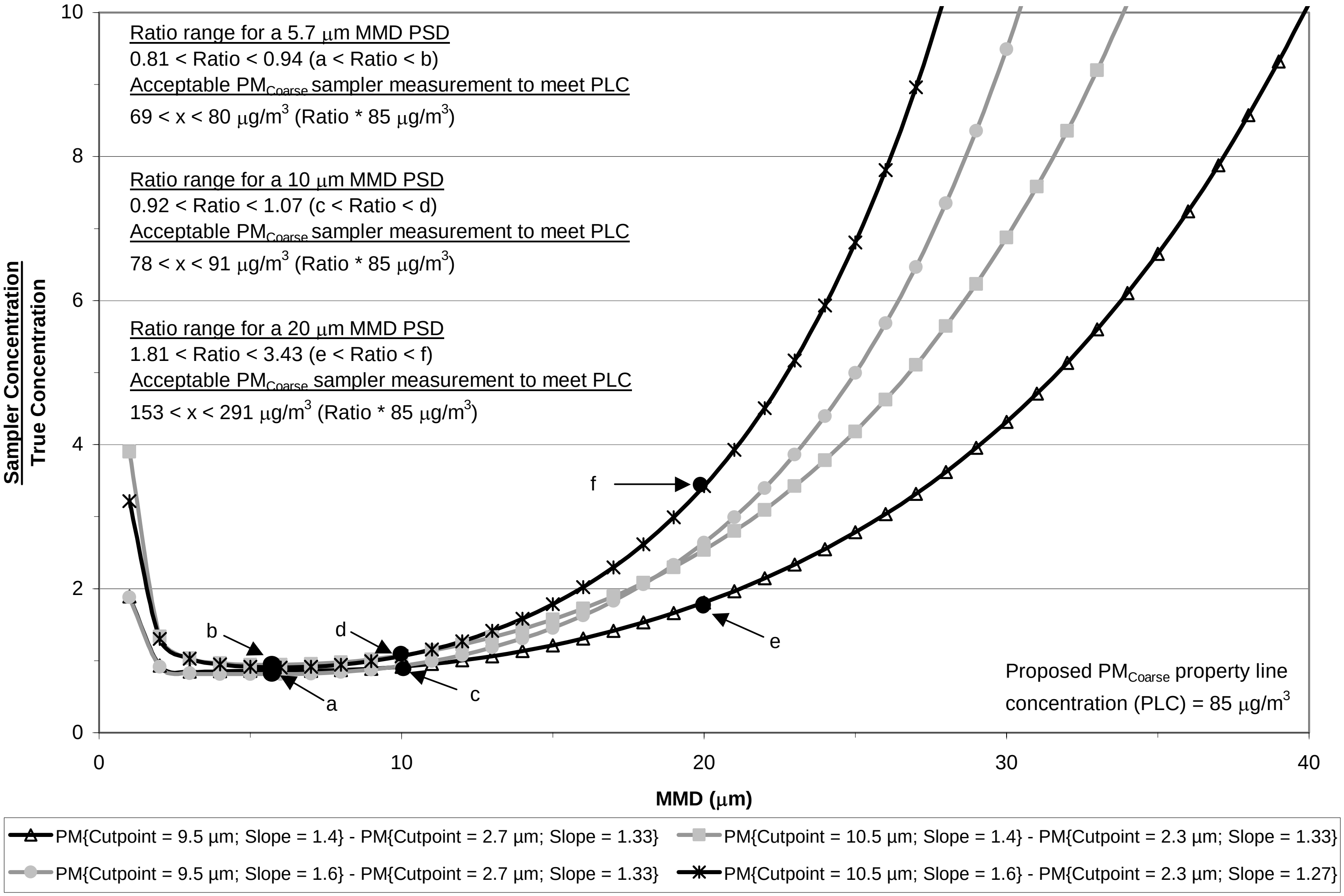


**Figure 41.** Theoretical ratios of $PM_{Coarse}$ to true PSD concentrations (PSD – GSD = 1.5).

In addition to using $PM_{10}$ and $PM_{2.5}$ sampler measurements to determine $PM_{coarse}$, many researchers and regulators are using the ratio of $PM_{2.5}$ to $PM_{10}$ sampler measurements to compare various data sets. A data set was compiled to examine the effects of the inherent errors attributed to the interaction of particle size and sampler performance characteristics on the ratios of $PM_{2.5}$ to $PM_{10}$ sampler concentrations. The $PM_{10}$ and $PM_{2.5}$ sampler performance characteristics used in the $PM_{coarse}$ evaluation were also used in the ratio of $PM_{2.5}$ to $PM_{10}$. The PSD characteristics were characterized by MMD values ranging from 1 to 40 μm and GSDs of 2.0 and 1.5. Results using a constant GSD of 2.0 and 1.5 are illustrated in Figures 42 and 43, respectively. The data presented in Figure 42 appears odd at first glance. For example, if the PSD were characterized by a MMD of 5.7 μm and a GSD of 2.0 then the ratio of sampler to true concentrations would range from 0.95 to 1.5. If the PSD were characterized by a MMD of 20 μm and a GSD of 2.0, then the ratio of sampler to true concentrations would range from 0.85 to 2.7. These ratios of sampler to true concentrations do not follow the general trends found for the $PM_{10}$ or $PM_{2.5}$ samplers. However when focusing on the equation for calculating the ratio of sampler to true concentrations for the ratio of $PM_{2.5}$ to $PM_{10}$, the results become logical. The following equation was used in the calculation

$$\frac{SamplerConcentration}{TrueConcentration} = \frac{\left(\dfrac{SamplerPM_{2.5}}{SamplerPM_{10}}\right)}{\left(\dfrac{TruePM_{2.5}}{TruePM_{10}}\right)} or \left(\frac{SamplerPM_{2.5}}{SamplerPM_{10}}\right) * \left(\frac{TruePM_{10}}{TruePM_{2.5}}\right) \quad (31)$$

In equation 31, when the MMD of the dust being sampled increases, with a constant GSD of 2.0, the ratio of true $PM_{10}$ to true $PM_{2.5}$ is increasing more rapidly than the ratio of sampler $PM_{2.5}$ to $PM_{10}$ is decreasing for the upper bound sampler performance characteristics. This observation is more clearly illustrated in Figure 42 for the curve representing the upper bound. The lower bound sampler performance characteristics cause the ratio of sampler $PM_{2.5}$ to $PM_{10}$ to decrease a little quicker than the ratio of true $PM_{10}$ to $PM_{2.5}$ is increasing for a GSD of 2.0. For a GSD of 1.5, the ratio of sampler $PM_{2.5}$ to $PM_{10}$ is decreasing a little slower than the ratio of true $PM_{10}$ to $PM_{2.5}$ is increasing. Based on these results, the ratio of $PM_{2.5}$ to $PM_{10}$ should not be used in comparing data sets.

## Effects of Sampler Performance Characteristics Varying Beyond Defined Tolerances

Another assumption associated with EPA approved samplers is that sampler performance characteristics are not affected by particle size characteristics or PM loadings. In other words, the performance characteristics associated with these samplers will remain within the EPA performance criteria regardless of particle size or PM loading. However, there are reports in the literature indicating that the performance characteristics of the EPA approved samplers do vary beyond the established criteria. For example, Wang et al. (2003) reported a $d_{50}$ of 13.2 μm and a slope of 1.25 for the Graseby-Anderson FRM $PM_{10}$ sampler when sampling a dust characterized by a MMD of 19 μm and a GSD of 1.4.

Based on the information in the literature, a series of calculations was performed in Mathcad 2000 to generate a data file containing the solutions to equations 25 and 30 over a range of parameters in order to evaluate the errors associated with sampler performance characteristics varying beyond established tolerances. The PSD parameters included in the calculations were limited to MMD values of 5.7, 10, and 20 μm and GSD values of 1.5 and 2.0. Simulations were conducted for the $PM_{10}$ and $PM_{2.5}$ ambient air samplers, calculation of $PM_{coarse}$, and the calculation of the ratio of $PM_{2.5}$ to $PM_{10}$. For the $PM_{10}$ ambient sampler simulations, the $d_{50}$ values ranged from 1 to 15 μm for slopes of 1.2, 1.5, 2.0, and 2.5. For the $PM_{2.5}$ ambient sampler simulations, the $d_{50}$ values ranged 1 to 10 μm for slopes of 1.2, 1.5, 2.0, and 2.5. For the $PM_{coarse}$ and ratio of $PM_{2.5}$ to $PM_{10}$ simulations, the $PM_{10}$ $d_{50}$ values ranged from 9 to 15 μm with a slope of 1.5 and the $PM_{2.5}$ $d_{50}$ values ranged from 1 to 6 μm with a slope of 1.3.

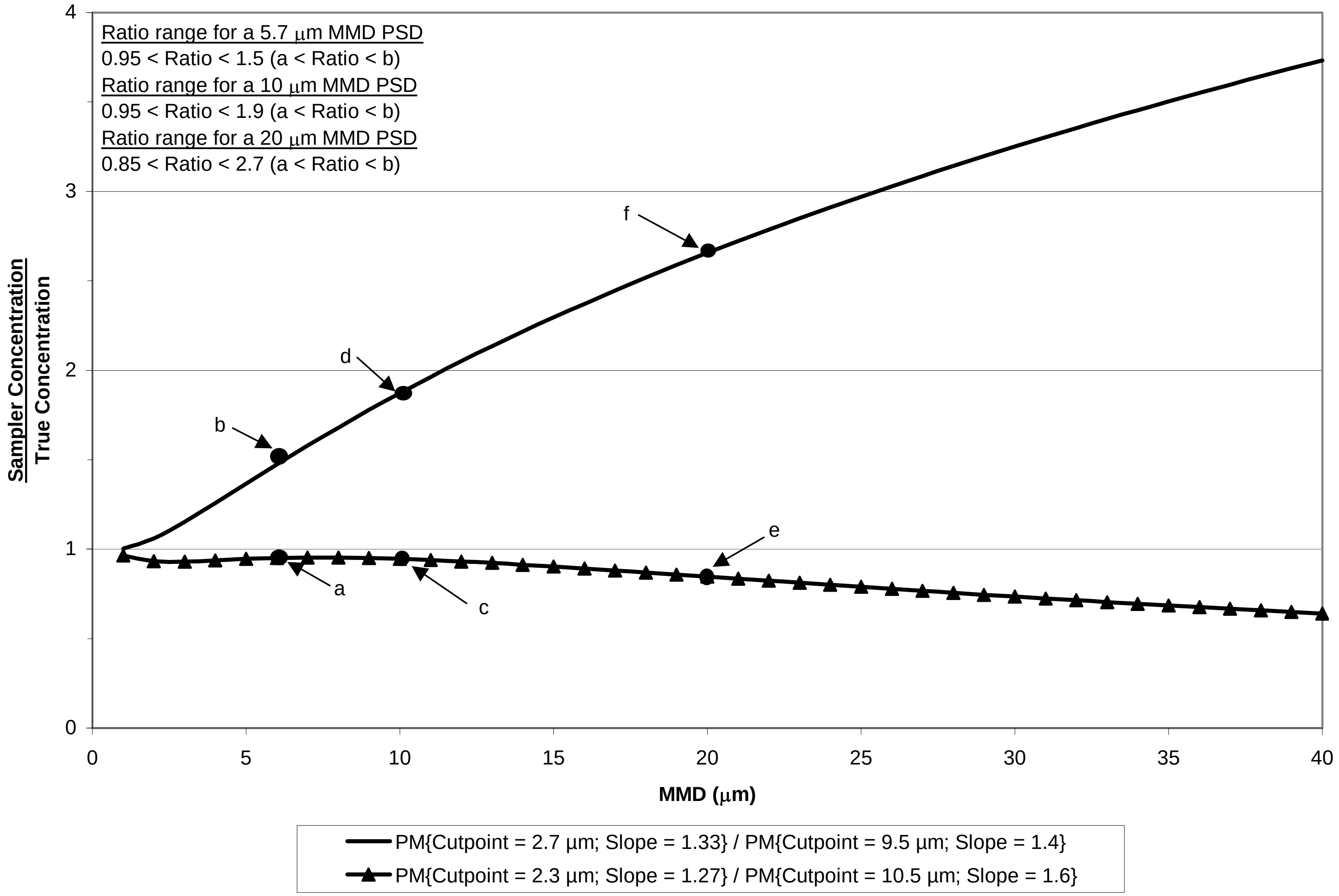


**Figure 42.** Theoretical ratios of $PM_{2.5/10}$ to true PSD concentrations (PSD – GSD = 2.0).

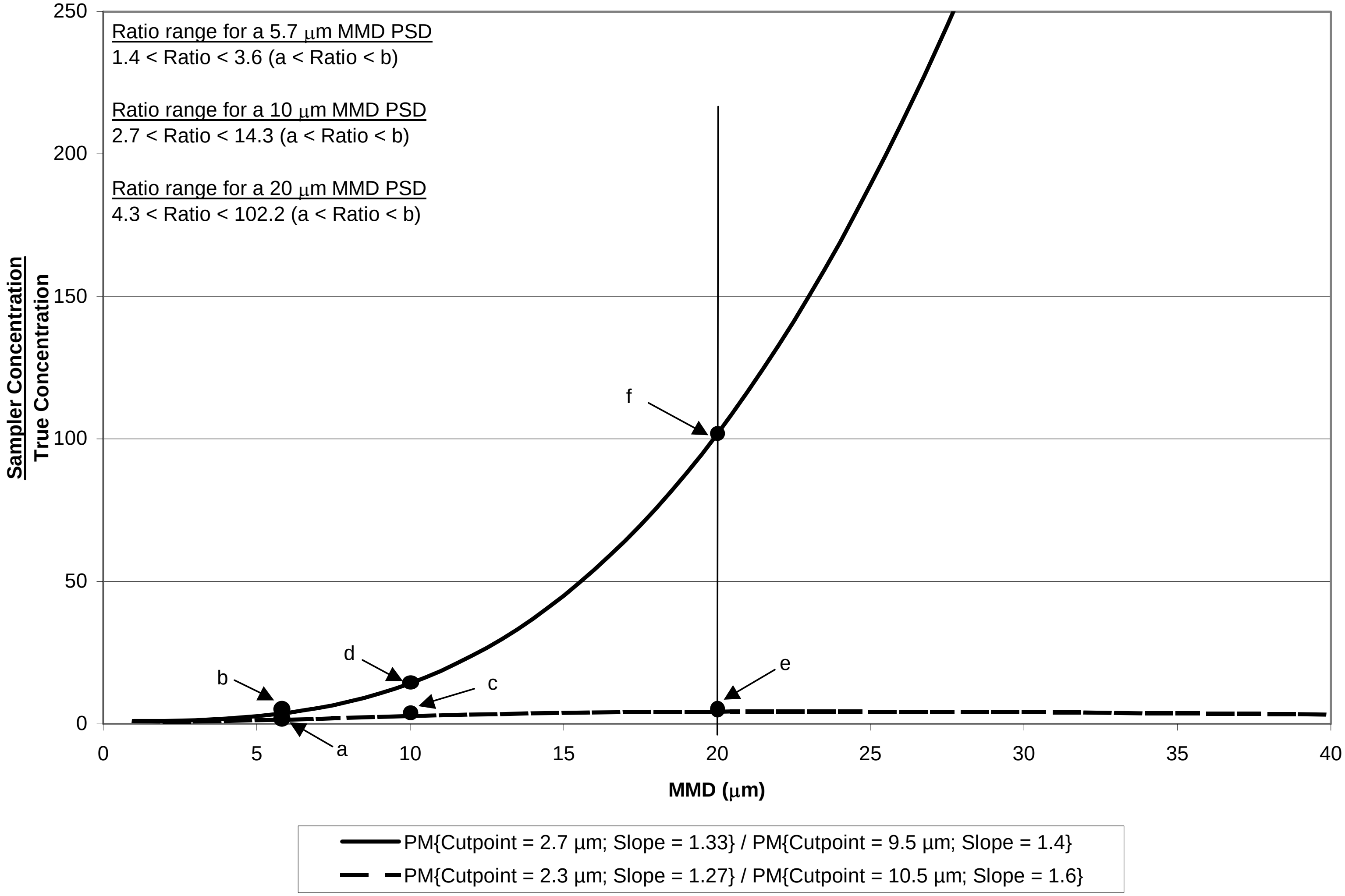


**Figure 43.** Theoretical ratios of $PM_{2.5/10}$ to true PSD concentrations (PSD – GSD = 1.5).

The data presented in Figures 44 through 48 correspond to the simulations for the $PM_{10}$ sampler. If a $PM_{10}$ sampler is exposed to a dust characterized by a MMD of 5.7 μm and a GSD of 2.25, as illustrated in Figure 44, and the $PM_{10}$ sampler performance characteristics were defined by a $d_{50}$ of 10 μm and a slope of 1.2 then the ratio of sampler to true concentrations would be 0.93. However, if the sampler performance characteristics were defined by a $d_{50}$ of 14 μm and a slope of 2.0 then the ratio of sampler to true concentration would increase to 1.14. A conclusion that can be drawn from the data illustrated in Figure 44 is that if the $PM_{10}$ sampler had a slope of 1.2, then the sampler's $d_{50}$ could be as high as 13 μm and the $PM_{10}$ sampler would still under-estimate the true $PM_{10}$ concentration.

If a $PM_{10}$ sampler is exposed to a dust characterized by a MMD of 10 μm and a GSD of 2.0, as illustrated in Figure 45, and the $PM_{10}$ sampler performance characteristics were defined by a $d_{50}$ of 10 μm and a slope of 1.2 (or for that matter any slope value) then the ratio of sampler to true concentrations would be 1.0. However, if the sampler performance characteristics were defined by a $d_{50}$ of 14 μm and a slope of 2.0 then the ratio of sampler to true concentration would increase to 1.36. If a $PM_{10}$ sampler is exposed to a dust characterized by a MMD of 10 μm and a GSD of 1.5, as illustrated in Figure 46, and the $PM_{10}$ sampler performance characteristics were defined by a $d_{50}$ of 10 μm and a slope of 1.2 (or for that matter any slope value) then the ratio of sampler to true concentrations would be 1.0. However, if the sampler performance characteristics were defined by a $d_{50}$ of 14 μm and a slope of 2.0 then the ratio of sampler to true concentration would increase to 1.55. A conclusion that can be drawn from the data illustrated in Figure 45 and 46 is that if the MMD of the dust being sampled is equal to the $d_{50}$ of the sampler then the sample will provide a true measure of $PM_{10}$, regardless of the samplers slope. Another conclusion that can be drawn from Figures 45 and 46 is that when the sampler $d_{50}$ is less than the MMD a larger sampler slope produces a larger ratio of sampler to true concentrations; however, if the sampler $d_{50}$ is larger than the MMD, then a smaller sampler slope will result in a larger ratio of sampler to true concentrations.

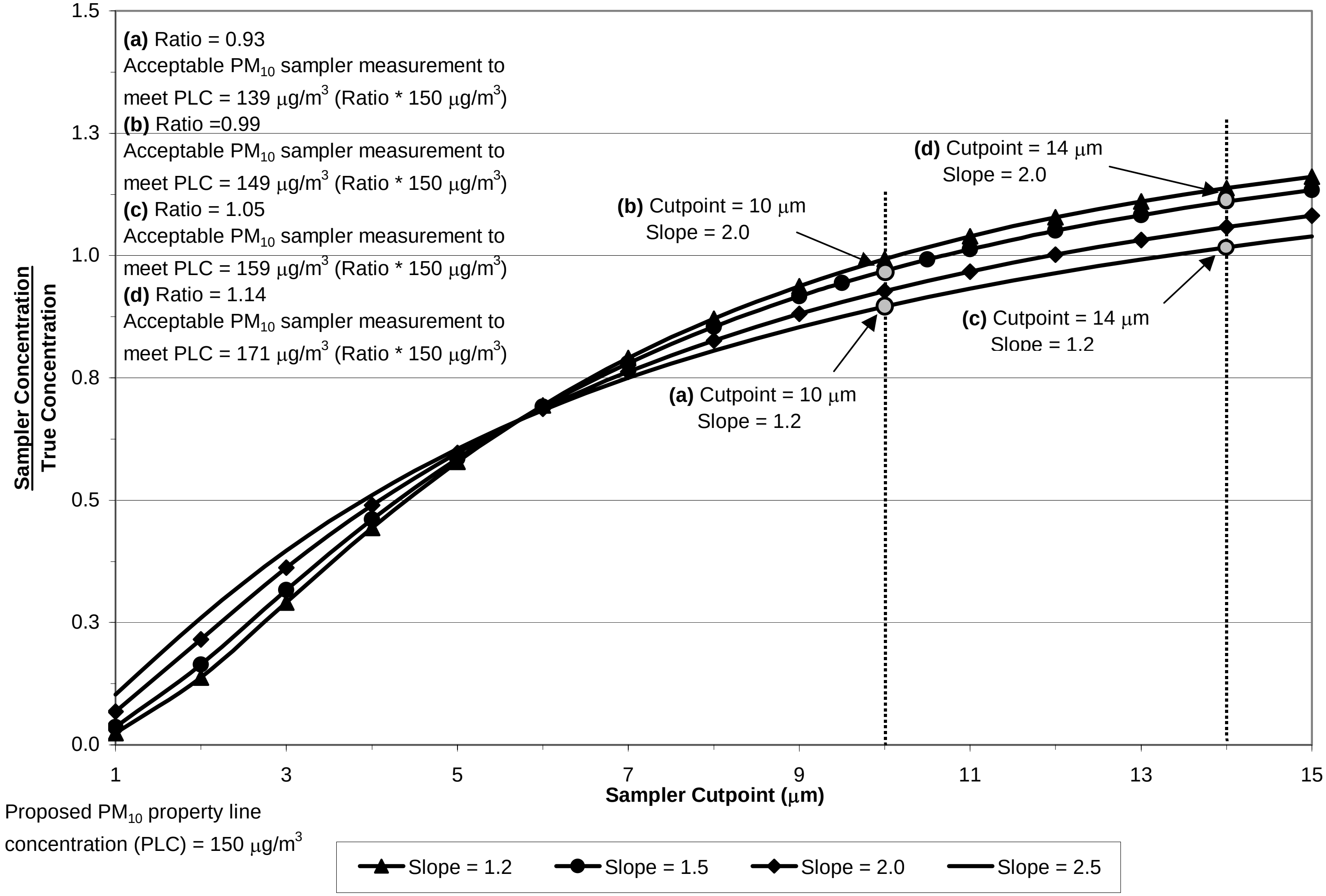


**Figure 44.** Effects of varying $PM_{10}$ sampler performance characteristics when theoretically exposed to a dust with a MMD of 5.7 μm and a GSD of 2.25.

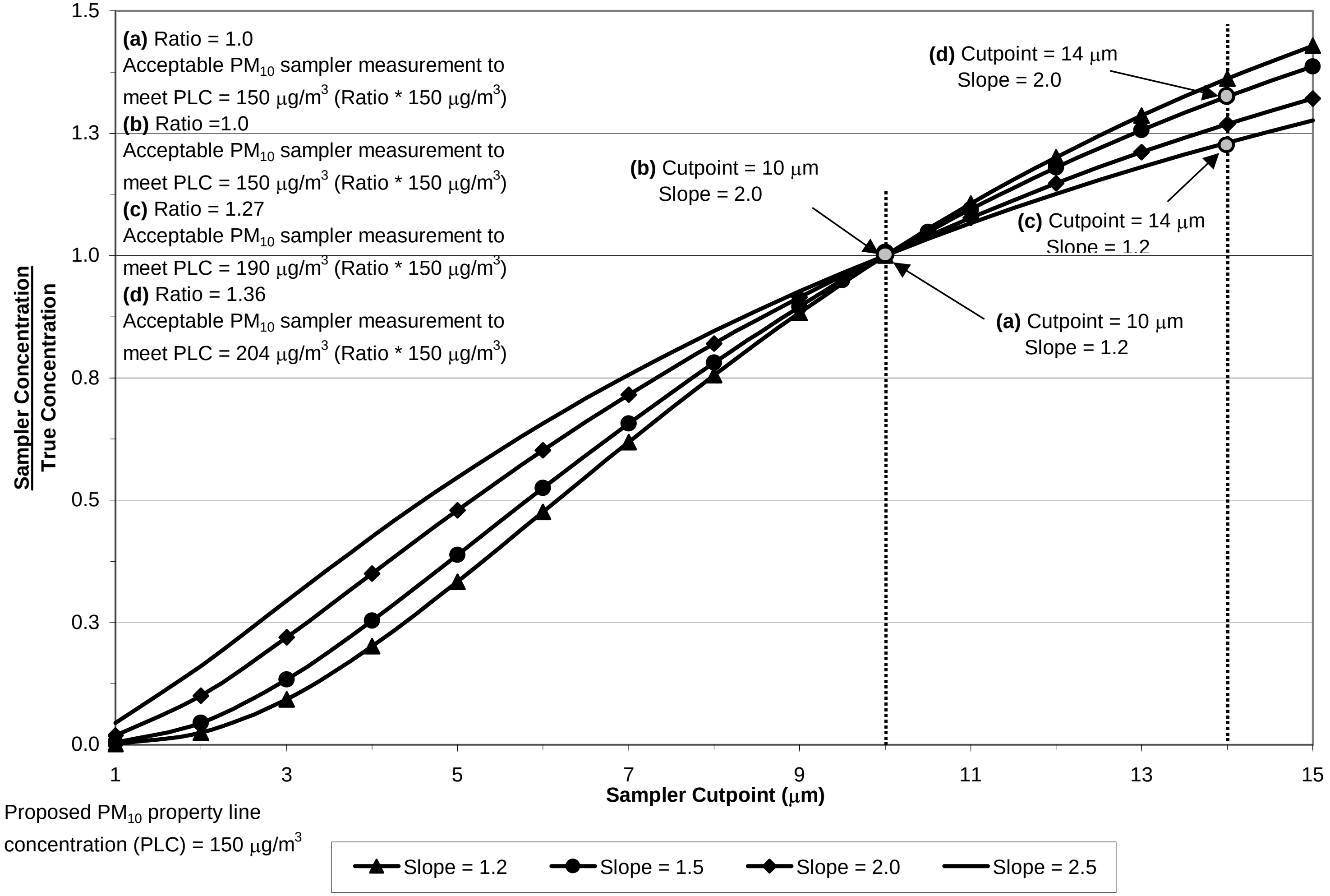


**Figure 45.** Effects of varying $PM_{10}$ sampler performance characteristics when theoretically exposed to a dust with a MMD of 10 μm and a GSD of 2.0.

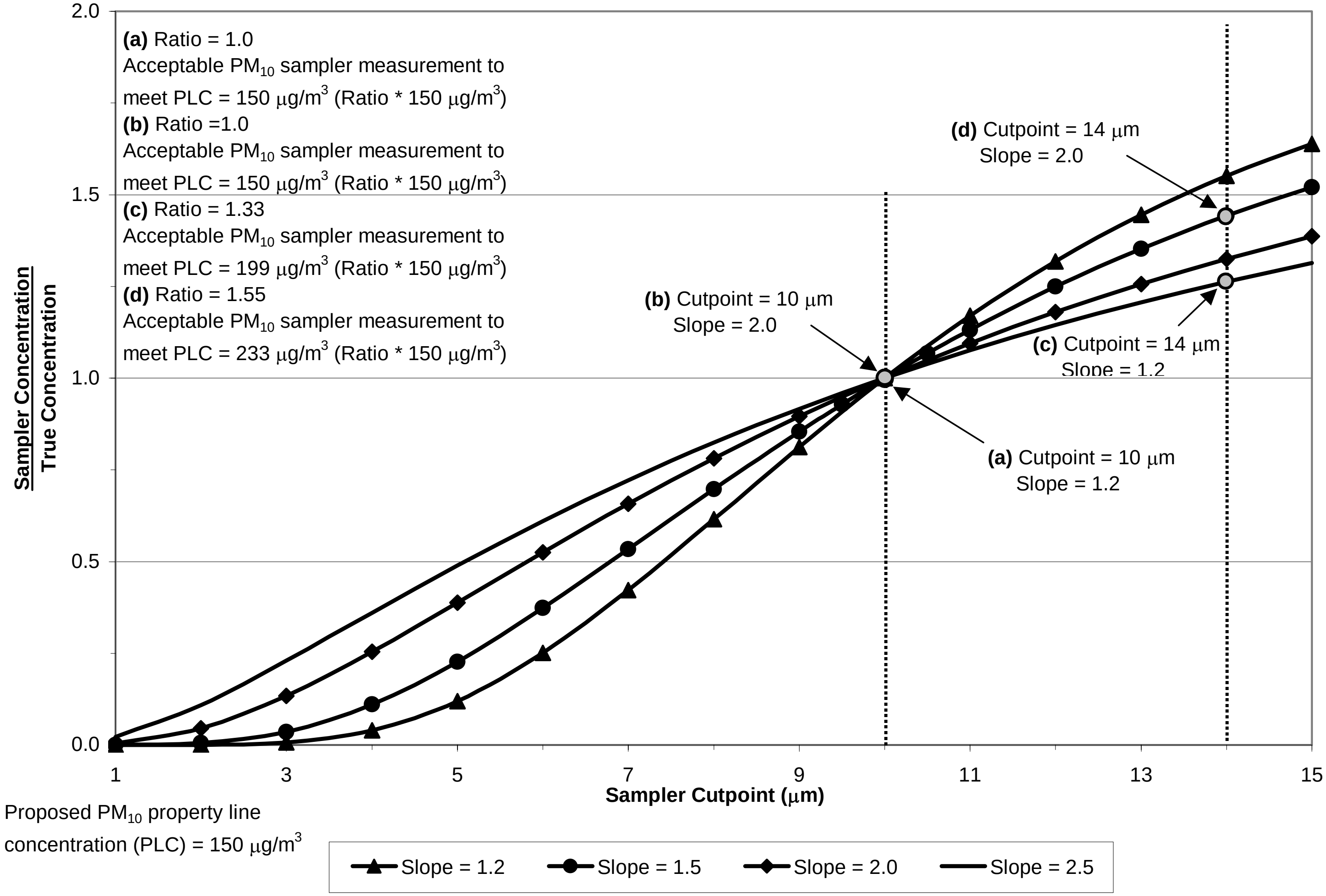


**Figure 46.** Effects of varying $PM_{10}$ sampler performance characteristics when theoretically exposed to a dust with a MMD of 10 μm and a GSD of 1.5.

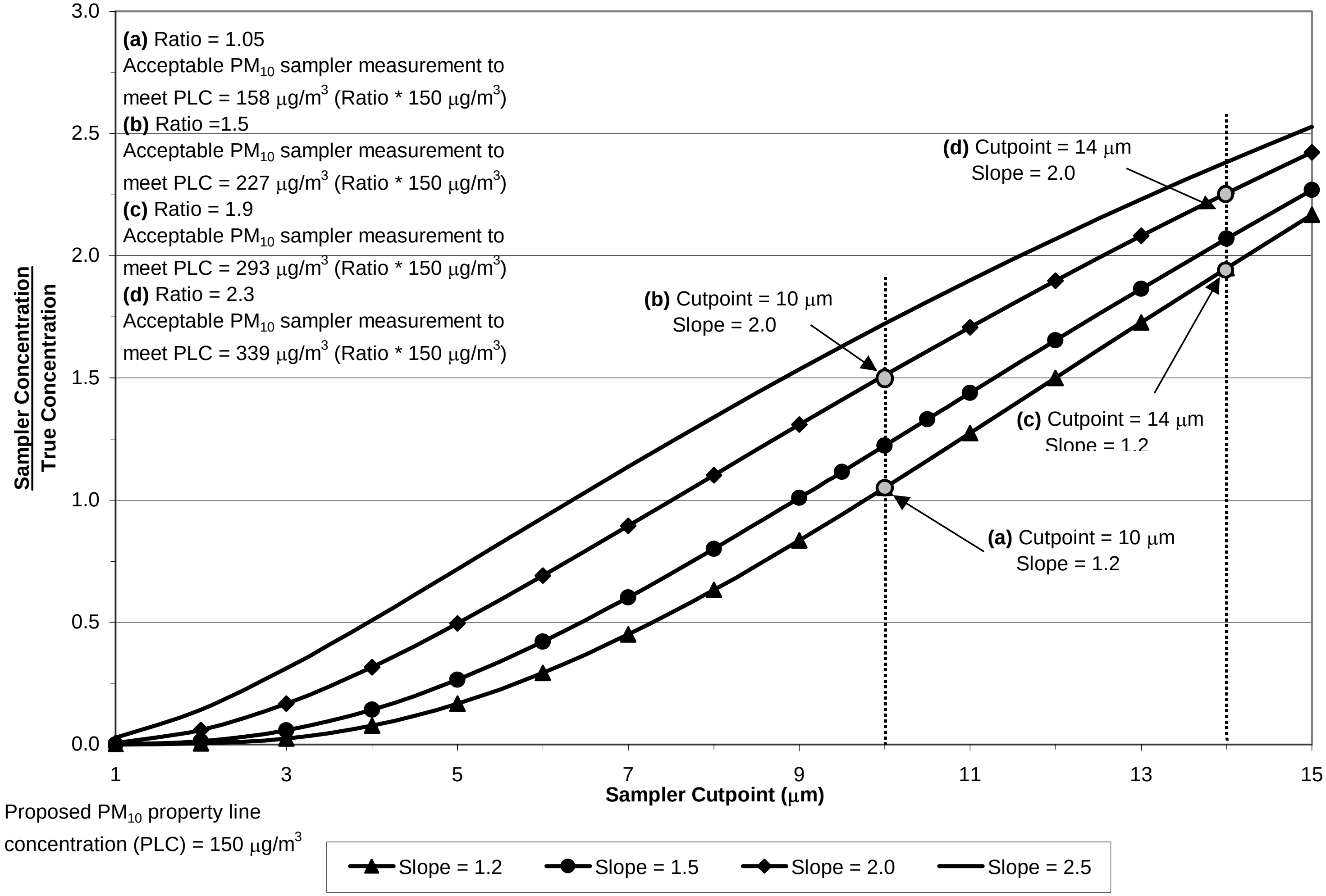


**Figure 47.** Effects of varying $PM_{10}$ sampler performance characteristics when theoretically exposed to a dust with a MMD of 20 μm and a GSD of 2.0.

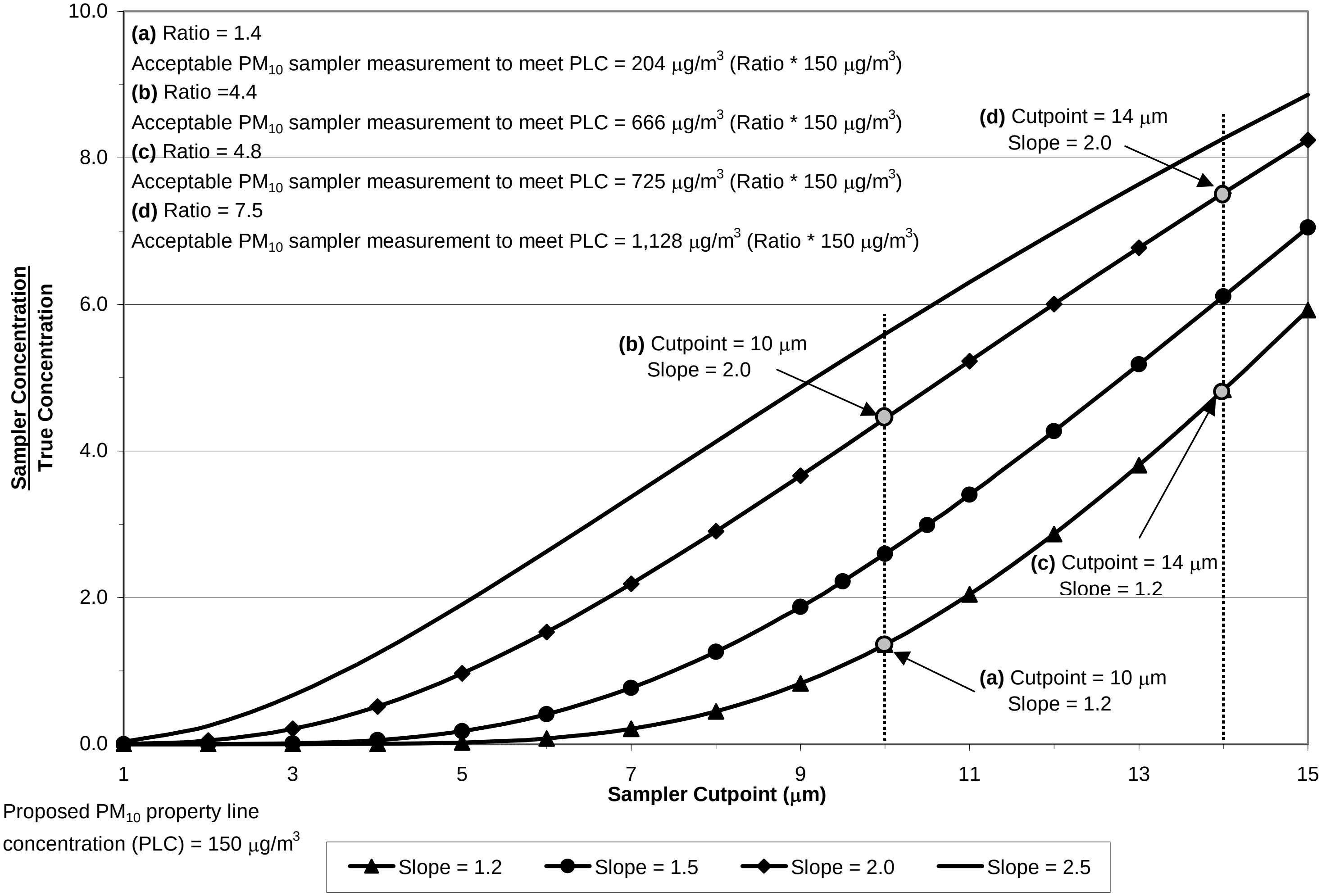
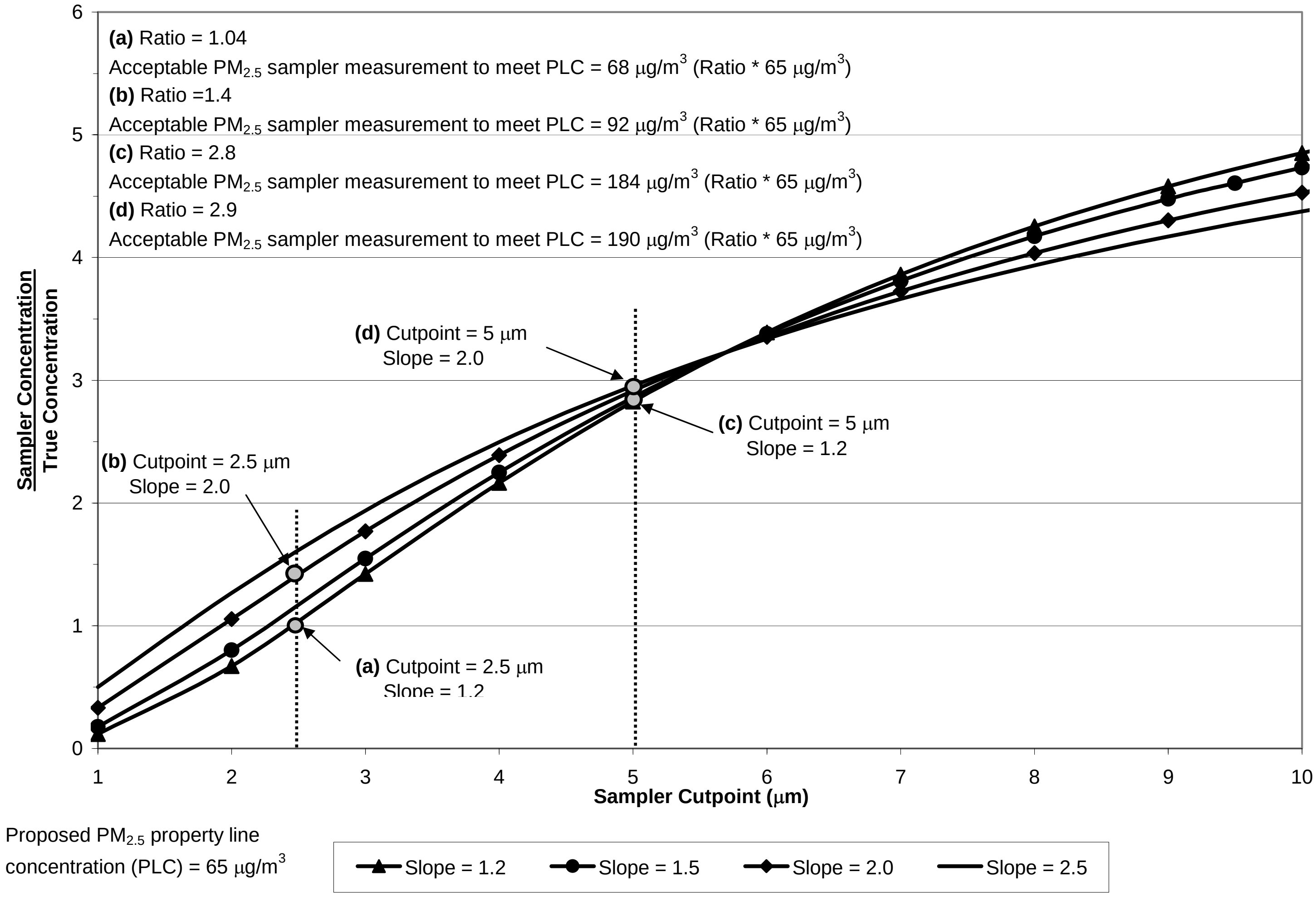


**Figure 48.** Effects of varying $PM_{10}$ sampler performance characteristics when theoretically exposed to a dust with a MMD of 20 µm and a GSD of 1.5.

If a $PM_{10}$ sampler is exposed to a dust characterized by a MMD of 20 μm and a GSD of 2.0, as illustrated in Figure 47, and the $PM_{10}$ sampler performance characteristics were defined by a $d_{50}$ of 10 μm and a slope of 1.2, then the ratio of sampler to true concentrations would be 1.05. However, if the sampler performance characteristics were defined by a $d_{50}$ of 14 μm and a slope of 2.0 then the ratio of sampler to true concentration would increase to 2.3.

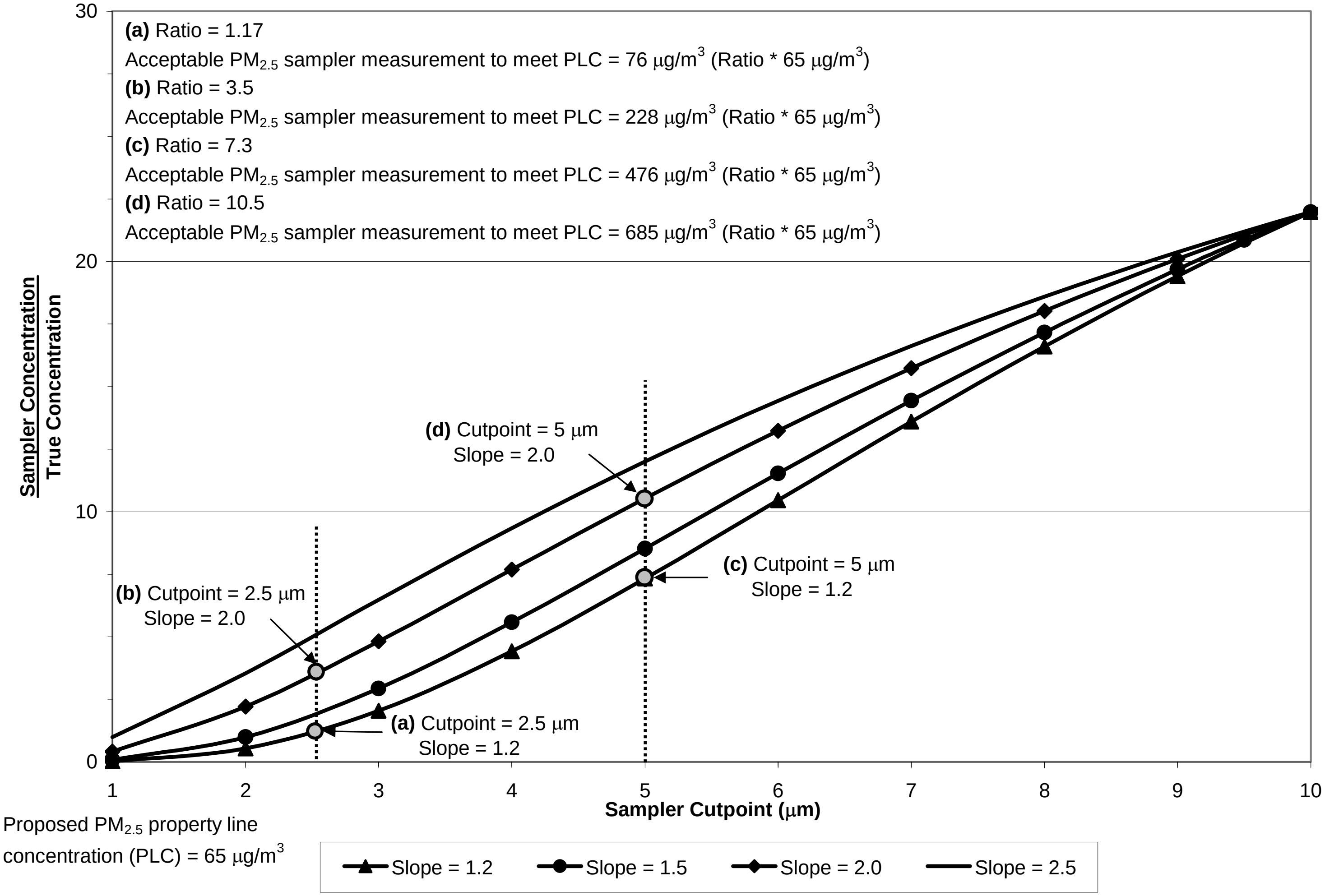
If a $PM_{10}$ sampler is exposed to a dust characterized by a MMD of 20 μm and a GSD of 1.5, as illustrated in Figure 48, and the $PM_{10}$ sampler performance characteristics were defined by a $d_{50}$ of 10 μm and a slope of 1.2, then the ratio of sampler to true concentrations would be 1.4. However, if the sampler performance characteristics were defined by a $d_{50}$ of 14 μm and a slope of 1.5, then the ratio of sampler to true concentration would increase to 7.5. A conclusion that can be drawn from the data illustrated in Figures 45 through 48 is that the $PM_{10}$ sampler slope impacts the ratio of sampler to true concentrations to a greater extent when the MMD of the dust being sampled is 20 μm, as compared to 10 μm.

The data presented in Figures 49 through 53 correspond to the simulations for the $PM_{2.5}$ sampler. If a $PM_{2.5}$ sampler is exposed to a dust characterized by a MMD of 5.7 μm and a GSD of 2.25, as illustrated in Figure 49, and the $PM_{2.5}$ sampler performance characteristics were defined by a $d_{50}$ of 2.5 μm and a slope of 1.2, then the ratio of sampler to true concentrations would be 1.04. However, if the sampler performance characteristics were defined by a $d_{50}$ of 5 μm and a slope of 2.0, then the ratio of sampler to true concentration would increase to 2.9.



Proposed $PM_{2.5}$ property line concentration (PLC) = 65 µg/$m^3$

**Figure 49.** Effects of varying $PM_{2.5}$ sampler performance characteristics when theoretically exposed to a dust with a MMD of 5.7 µm and a GSD of 2.25.



**(a)** Ratio = 1.17
Acceptable $PM_{2.5}$ sampler measurement to meet PLC = 76 μg/$m^3$ (Ratio * 65 μg/$m^3$)
**(b)** Ratio = 3.5
Acceptable $PM_{2.5}$ sampler measurement to meet PLC = 228 μg/$m^3$ (Ratio * 65 μg/$m^3$)
**(c)** Ratio = 7.3
Acceptable $PM_{2.5}$ sampler measurement to meet PLC = 476 μg/$m^3$ (Ratio * 65 μg/$m^3$)
**(d)** Ratio = 10.5
Acceptable $PM_{2.5}$ sampler measurement to meet PLC = 685 μg/$m^3$ (Ratio * 65 μg/$m^3$)

**(d)** Cutpoint = 5 μm
Slope = 2.0

**(c)** Cutpoint = 5 μm
Slope = 1.2

**(b)** Cutpoint = 2.5 μm
Slope = 2.0

**(a)** Cutpoint = 2.5 μm
Slope = 1.2

Proposed $PM_{2.5}$ property line
concentration (PLC) = 65 μg/$m^3$

**Figure 50.** Effects of varying $PM_{2.5}$ sampler performance characteristics when theoretically exposed to a dust with a MMD of 10 μm and a GSD of 2.0.

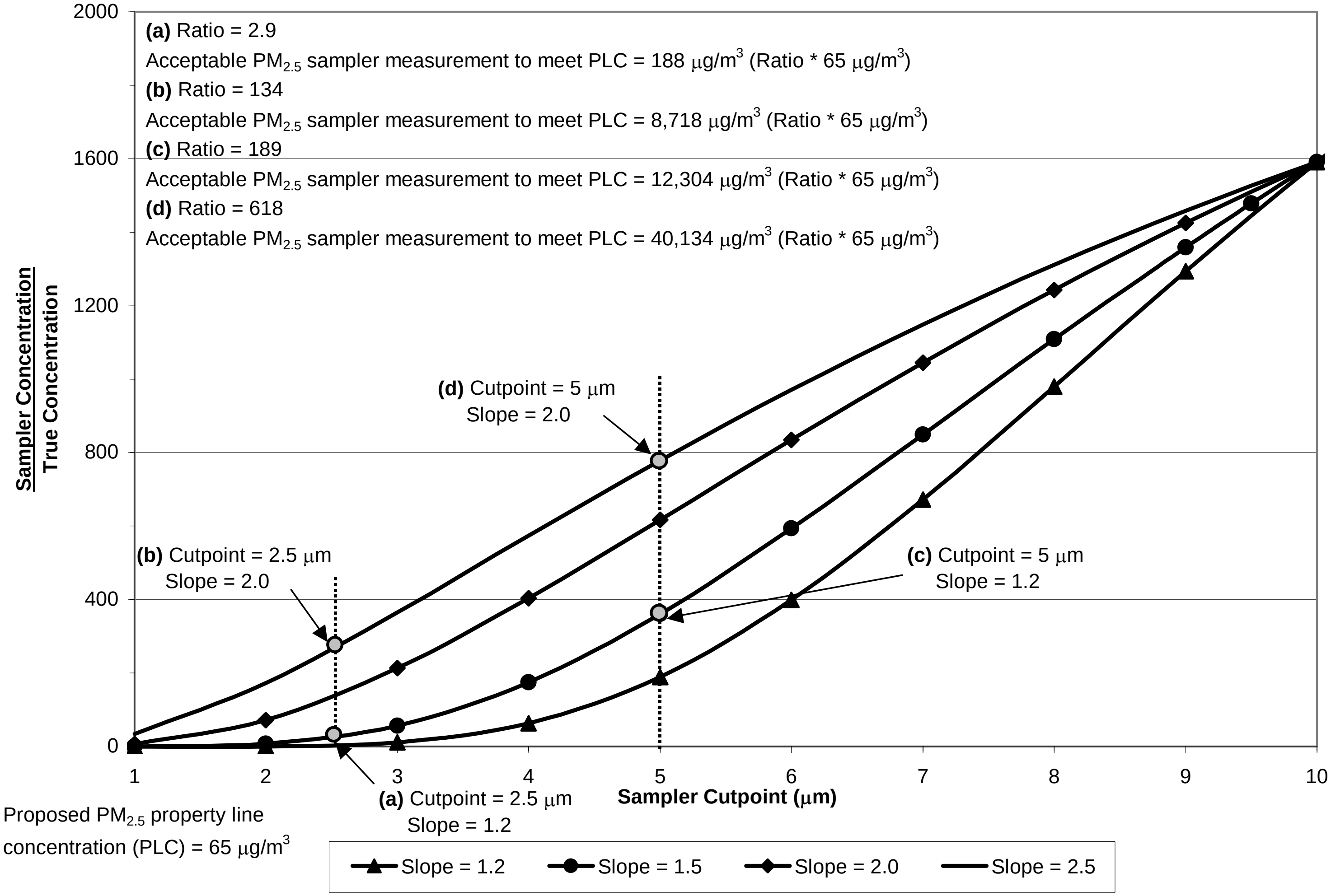


**Figure 51.** Effects of varying $PM_{2.5}$ sampler performance characteristics when theoretically exposed to a dust with a MMD of 10 μm and a GSD of 1.5.

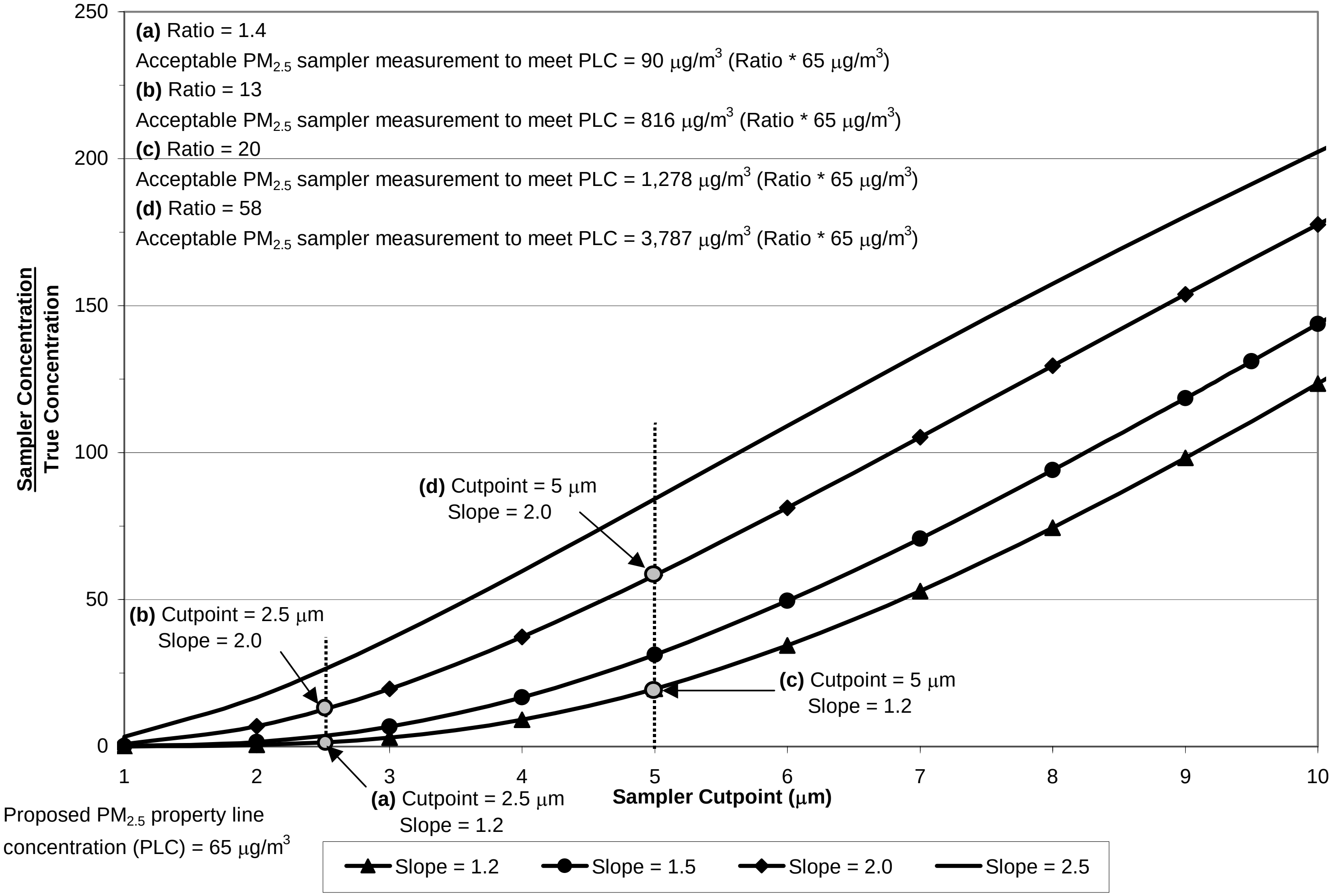


**Figure 52.** Effects of varying $PM_{2.5}$ sampler performance characteristics when theoretically exposed to a dust with a MMD of 20 μm and a GSD of 2.0.

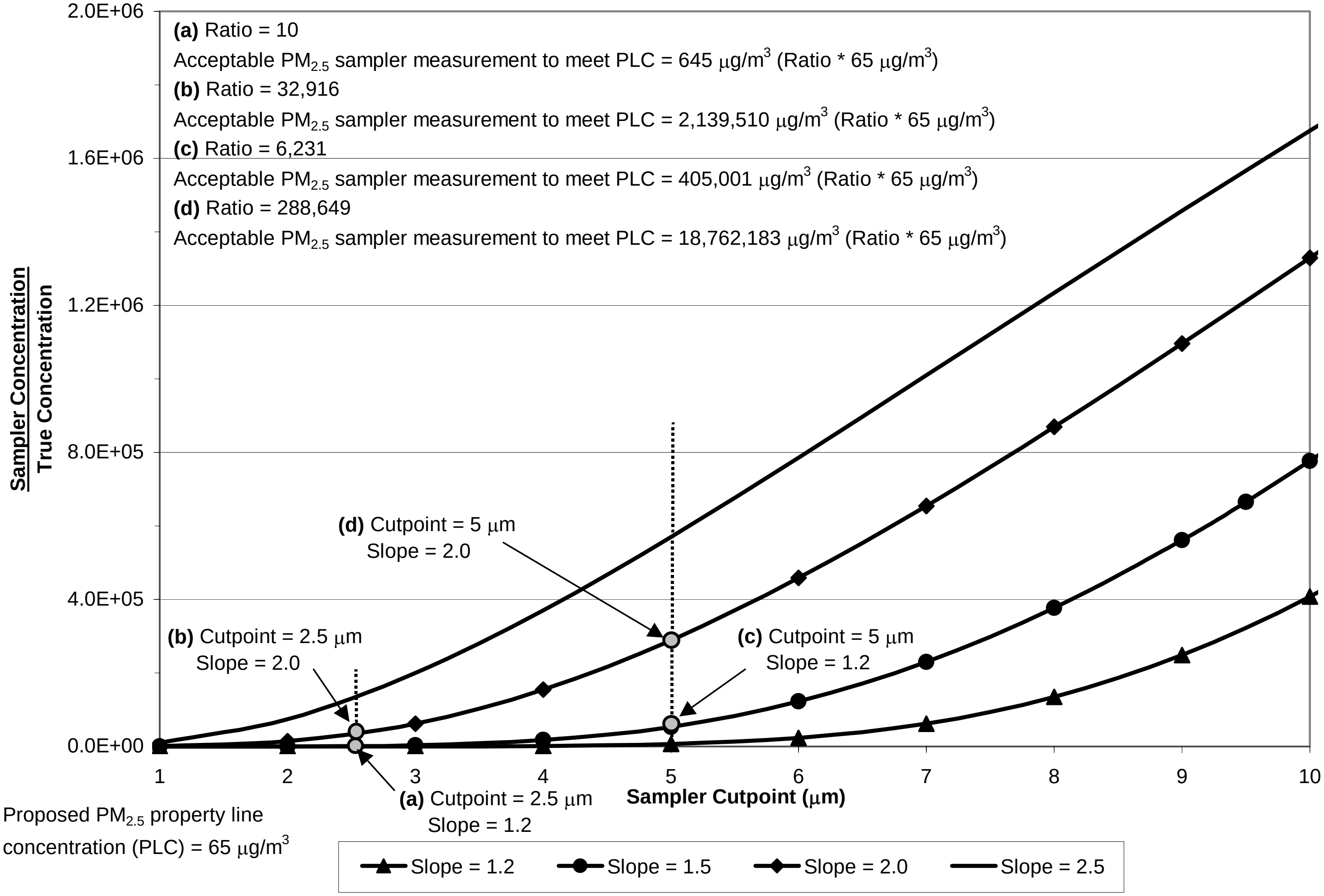


**Figure 53.** Effects of varying $PM_{2.5}$ sampler performance characteristics when theoretically exposed to a dust with a MMD of 20 μm and a GSD of 1.5.

If a $PM_{2.5}$ sampler is exposed to a dust characterized by a MMD of 10 μm and a GSD of 2.0, as illustrated in Figure 50, and the $PM_{2.5}$ sampler performance characteristics were defined by a $d_{50}$ of 2.5 μm and a slope of 1.2, then the ratio of sampler to true concentrations would be 1.17. However, if the sampler performance characteristics were defined by a $d_{50}$ of 5 μm and a slope of 2.0, then the ratio of sampler to true concentration would increase to 10.5. If a $PM_{2.5}$ sampler is exposed to a dust characterized by a MMD of 10 μm and a GSD of 1.5, as illustrated in Figure 51, and the $PM_{2.5}$ sampler performance characteristics were defined by a $d_{50}$ of 2.5 μm and a slope of 1.2, then the ratio of sampler to true concentrations would be 2.9. However, if the sampler performance characteristics were defined by a $d_{50}$ of 5 μm and a slope of 2.0 then the ratio of sampler to true concentration would increase to 618.

If a $PM_{2.5}$ sampler is exposed to a dust characterized by a MMD of 20 μm and a GSD of 2.0, as illustrated in Figure 52, and the $PM_{2.5}$ sampler performance characteristics were defined by a $d_{50}$ of 2.5 μm and a slope of 1.2, then the ratio of sampler to true concentrations would be 1.4. However, if the sampler performance characteristics were defined by a $d_{50}$ of 5 μm and a slope of 2.0, then the ratio of sampler to true concentration would increase to 58. If a $PM_{2.5}$ sampler is exposed to a dust characterized by a MMD of 20 μm and a GSD of 1.5, as illustrated in Figure 53, and the $PM_{2.5}$ sampler performance characteristics were defined by a $d_{50}$ of 2.5 μm and a slope of 1.2, then the ratio of sampler to true concentrations would be 10. However, if the sampler performance characteristics were defined by a $d_{50}$ of 5 μm and a slope of 2.0 then the ratio of sampler to true concentration would increase to 288,649. A conclusion that can be drawn from Figures 44 through 53 is that an increase in the sampler $d_{50}$ or slope impacts the ratio of sampler to true concentrations to a greater extent for dusts characterized by a MMD of 20 μm than a dust with an MMD of 10 μm. When a sampler's penetration curve shifts to the right, beyond the establish tolerances, the inherent errors associated with the samplers are greatly increased.

A series of five graphs were generated to illustrate the effects of varying sampler performance characteristics on measurements of $PM_{coarse}$. Figures 54 through 58 correspond to PSDs characterized by a MMD of 5.7 μm with a GSD of 2.25, MMD of 10 μm with a GSD of 2.0, MMD of 10 μm with a GSD of 1.5, MMD of 20 μm with a GSD of 2.0, and a MMD of 20 μm with a GSD of 1.5, respectively. General conclusions that can be drawn from Figures 54 through 58 are: 1) an increase in the $PM_{2.5}$ $d_{50}$ value will decrease the ratio of sampler to true concentrations (i.e. the increase in the $PM_{2.5}$ $d_{50}$ will help offset the error associated with the $PM_{10}$ sampler); 2) a decrease in the GSD will result in an increase in the ratio of sampler to true concentrations; 3) as the MMD increases, the ratio of sampler to true concentrations increases; 4) as the $PM_{10}$ sampler $d_{50}$ is increased the ratio of sampler to true concentrations increases; and 5) an increase in the $PM_{10}$ sampler $d_{50}$ has a greater effect than an increase in the $PM_{2.5}$ sampler $d_{50}$ when the MMD is greater than 10 μm.

A series of five graphs were generated to illustrate the effects of varying sampler performance characteristics on the calculations of the ratio of $PM_{2.5}$ to $PM_{10}$. Figures 59 through 63 correspond to PSDs characterized by a MMD of 5.7 μm with a GSD of 2.25, MMD of 10 μm with a GSD of 2.0, MMD of 10 μm with a GSD of 1.5, MMD of 20 μm with a GSD of 2.0, and a MMD of 20 μm with a GSD of 1.5, respectively. General conclusions that can be drawn from Figures 59 through 63 are: 1) an increase in the $PM_{2.5}$ sampler $d_{50}$ results in an increase in the ratio of sampler to true concentration; 2) an increase in the $PM_{10}$ sampler $d_{50}$ results in a decrease in the ratio of sampler to true concentrations; 3) the $PM_{2.5}$ $d_{50}$ has a larger impact on the ratio of sampler to true concentrations as compared to the $PM_{10}$ $d_{50}$; 4) for a MMD of 20 μm and a GSD of 1.5, a decrease in the $PM_{10}$ $d_{50}$ from 10 μm to 9 μm greatly increases the ratio of sampler to true concentrations; 5) an increase in the MMD results in an increase in the ratio of sampler to true concentrations; and 6) a decrease in GSD results in an increase in the ratio of sampler to true concentrations.

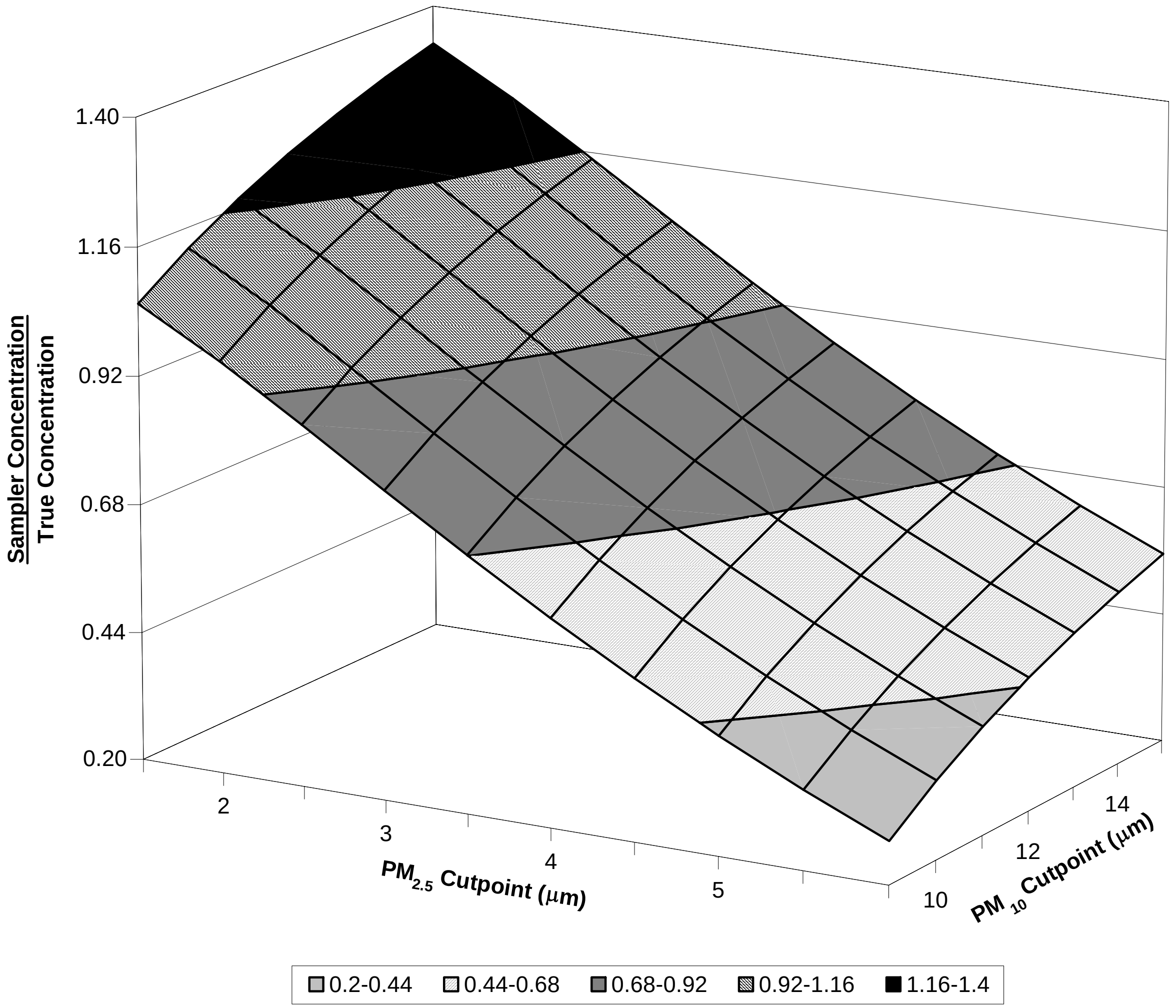


**Figure 54.** Theoretical ratios of $PM_{Coarse}$ to true PSD concentrations for varying $PM_{2.5}$ and $PM_{10}$ sampler cutpoints (PSD – MMD = 5.7; GSD = 2.25; $PM_{2.5}$ sampler slope = 1.3; $PM_{10}$ sampler slope = 1.5).

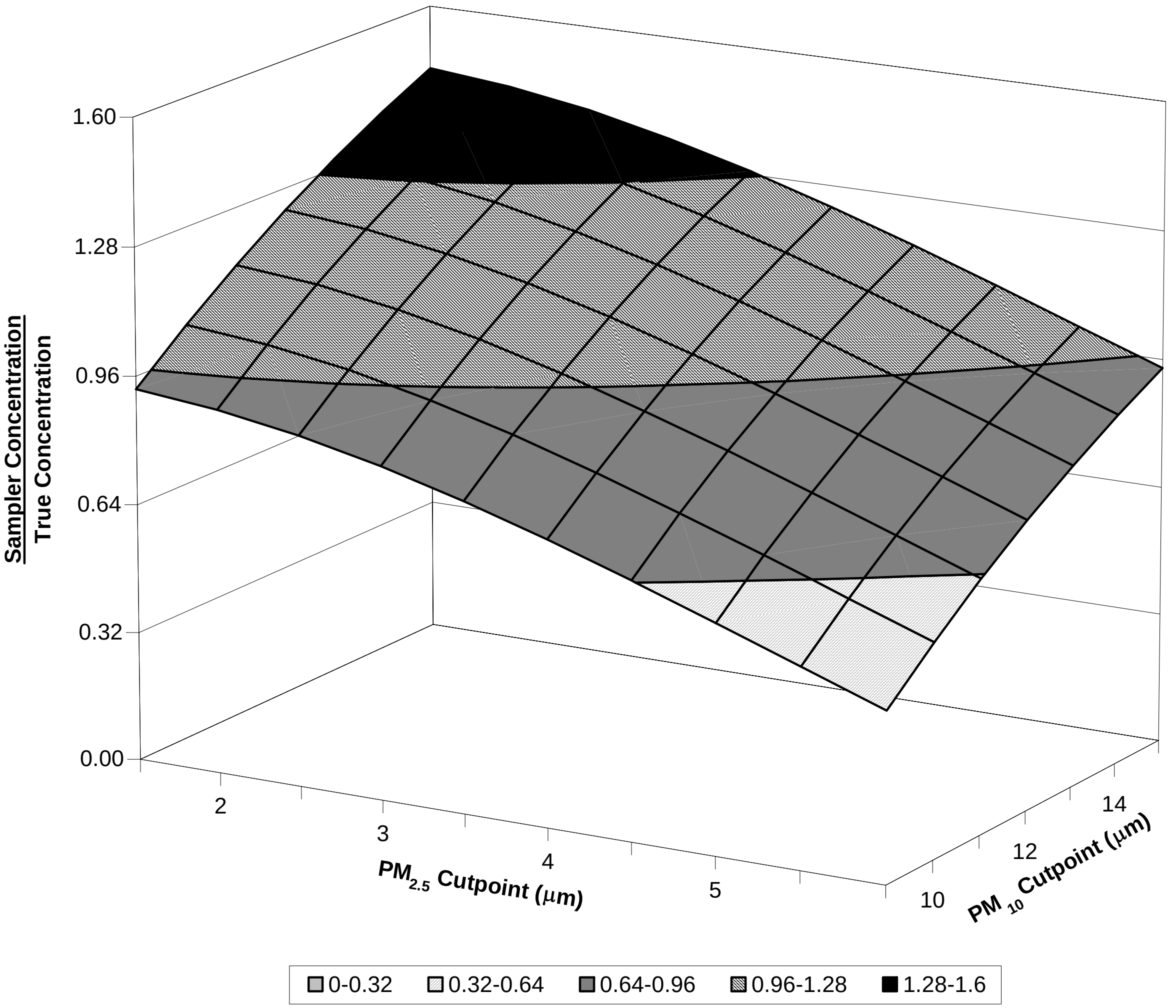


**Figure 55.** Theoretical ratios of PM$_{\text{Coarse}}$ to true PSD concentrations for varying PM$_{2.5}$ and PM$_{10}$ sampler cutpoints (PSD – MMD = 10; GSD = 2.0; PM$_{2.5}$ sampler slope = 1.3; PM$_{10}$ sampler slope = 1.5).

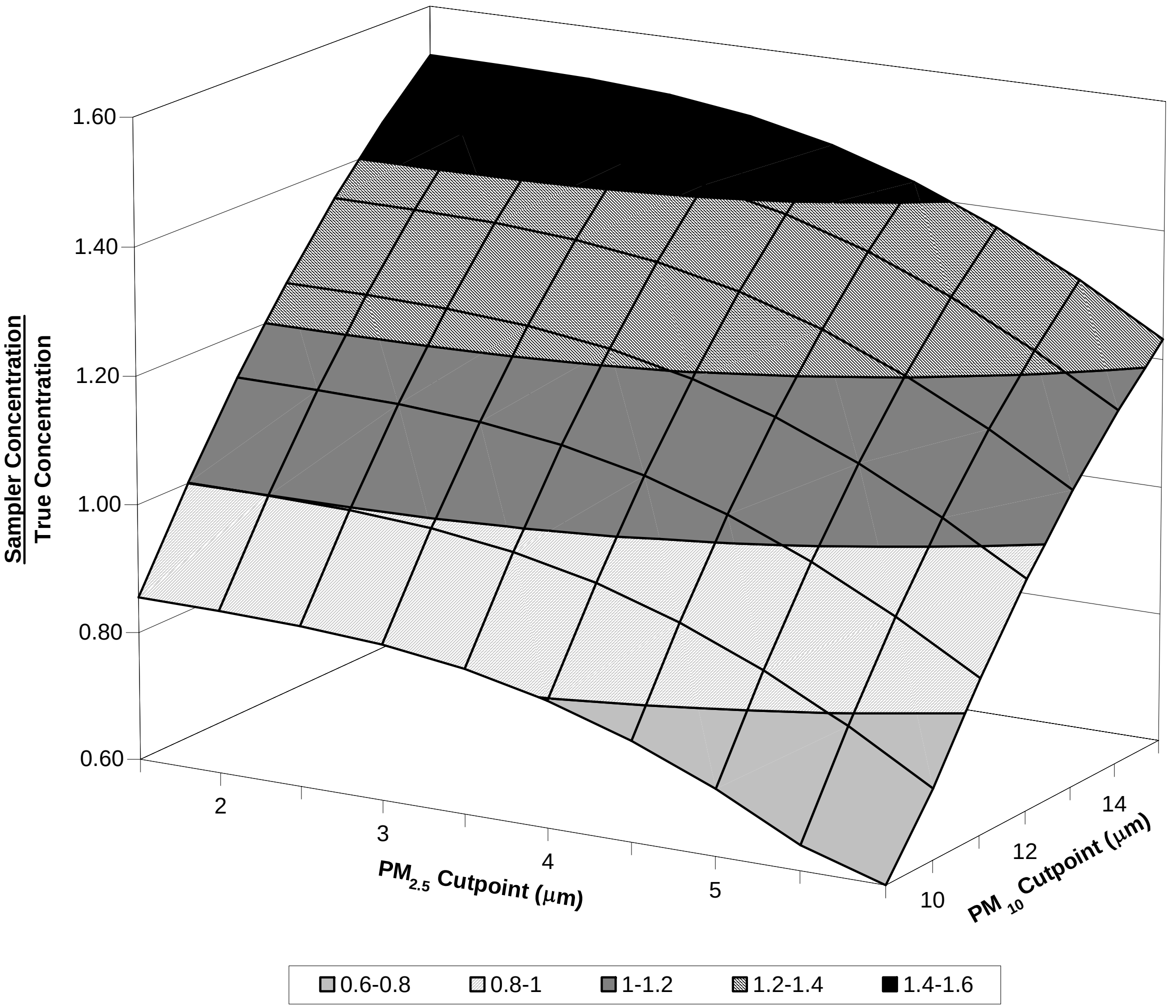


**Figure 56.** Theoretical ratios of $PM_{Coarse}$ to true PSD concentrations for varying $PM_{2.5}$ and $PM_{10}$ sampler cutpoints (PSD – MMD = 10; GSD = 1.5; $PM_{2.5}$ sampler slope = 1.3; $PM_{10}$ sampler slope = 1.5).

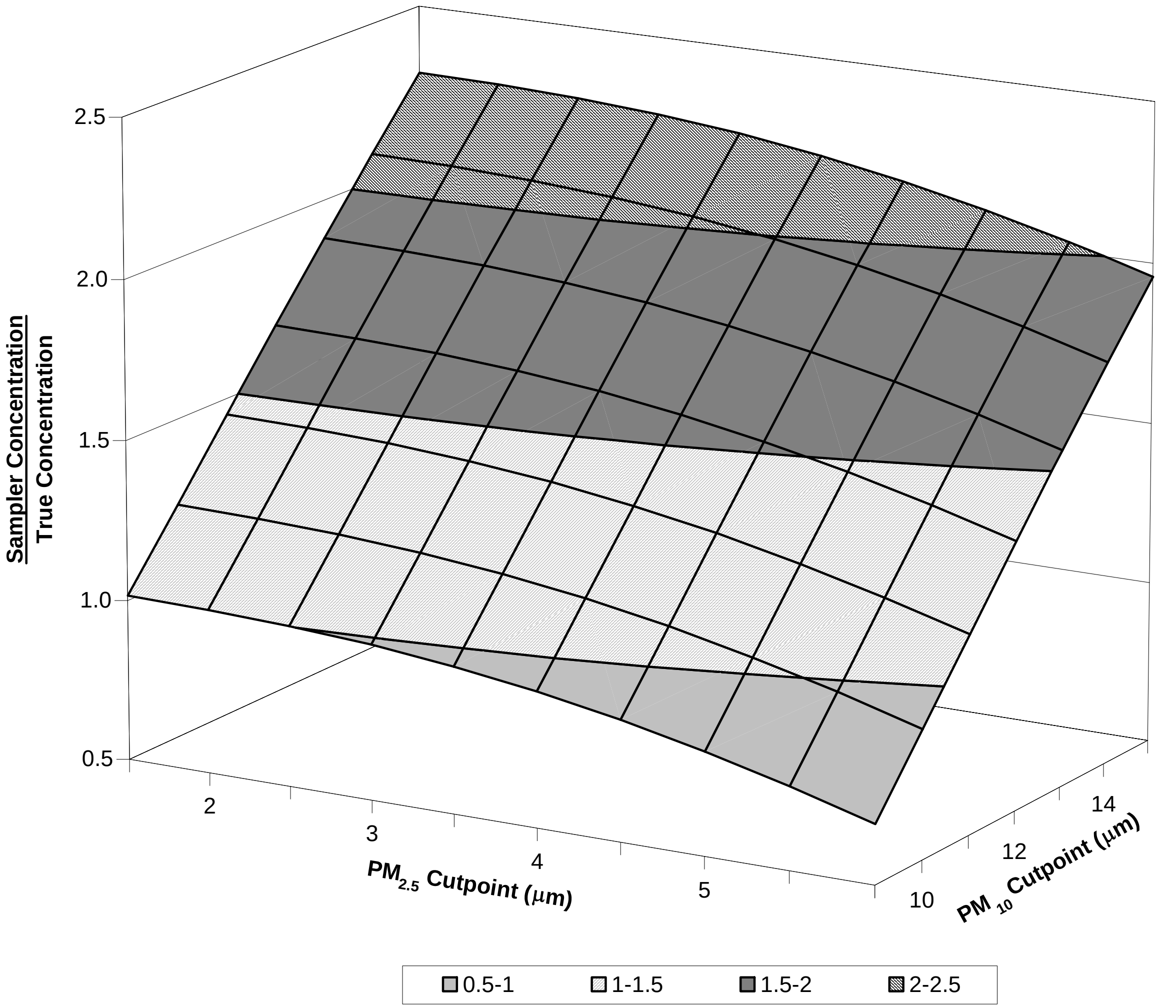


**Figure 57.** Theoretical ratios of $PM_{Coarse}$ to true PSD concentrations for varying $PM_{2.5}$ and $PM_{10}$ sampler cutpoints (PSD – MMD = 20; GSD = 2.0; $PM_{2.5}$ sampler slope = 1.3; $PM_{10}$ sampler slope = 1.5).

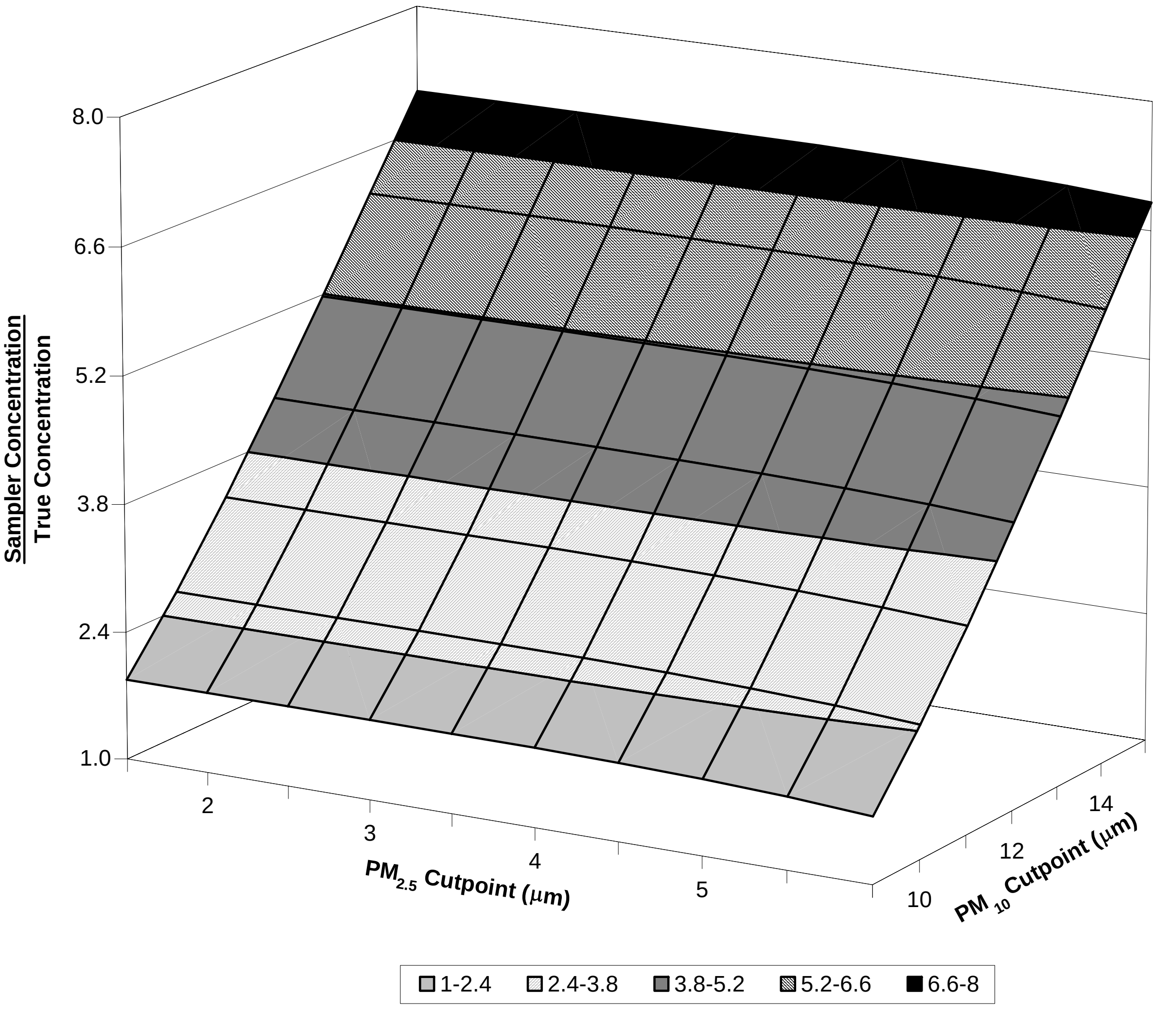


**Figure 58.** Theoretical ratios of $PM_{Coarse}$ to true PSD concentrations for varying $PM_{2.5}$ and $PM_{10}$ sampler cutpoints (PSD – MMD = 20; GSD = 1.5; $PM_{2.5}$ sampler slope = 1.3; $PM_{10}$ sampler slope = 1.5).

In order to further explore the effects of sampler performance characteristics on the inherent errors associated with PM sampling, Mathcad 2000 was used to generate a data set corresponding to various sampler performance characteristics for a limited range of sampler performance characteristics. For this data set, the $d_{50}$ used in equations 18 and 20 were set equal to each other. In other words, the data set is used to examine if a samplers $d_{50}$ is impacted by various PSD characteristics. The sampler slopes evaluated included: 1.1, 1.2, 1.3, 1.5, 2.0, and 2.5. The PSDs were characterized as a MMD of 5.7 μm with a GSD of 2.25, MMD of 10 μm with a GSD of 2.0, MMD of 10 μm with a GSD of 1.5, MMD of 20 μm with a slope of 2.0, and a MMD of 20 μm with a GSD of 1.5 for Figures 64 through 68, respectively. If the PSD were characterized by a MMD of 5.7 μm and a GSD of 2.25, as illustrated in Figure 64, a PM sampler with a $d_{50}$ greater than about 4.5 μm will results in sampler concentration very similar to the true concentrations. If the PM sampler possessed a slope of 1.2 or smaller, the sampler concentration would be very similar to the true concentration. In general, the EPA approved $PM_{2.5}$ and $PM_{10}$ samplers will work fairly well in urban environments.

If the PSD were characterized by a MMD of 10 μm and a GSD of 2.0, as illustrated in Figure 65, a PM sampler with a $d_{50}$ greater than about 9 μm will results in sampler concentration very similar to the true concentrations. If the PM sampler possessed a slope of 1.2 or smaller, the ratio of sampler to true concentrations would be 1.2 or smaller if the sampler $d_{50}$ is greater than 2.5 μm. If the PSD were characterized by a MMD of 10 μm and a GSD of 1.5, as illustrated in Figure 66, the ratio of sampler to true concentrations is greatly impacted in sampler $d_{50}$'s varying much beyond 10 μm in either direction. A PM sampler slope of 1.1 and a $d_{50}$ of 2.5 μm results in a ratio of sampler to true concentrations of 1.3. The results illustrated in Figures 67 and 68 are similar to those in Figures 65 and 66, except the increase in the ratio of sampler to true concentration becomes more pronounced when the MMD is increased from 10 μm to 20 μm. In general, the errors associated with PM samplers with $d_{50}$ values of 10 μm or less will be substantial for PM characterized with a MMD of 10 μm or larger (i.e. the errors will increase as the MMD of the dust increases).

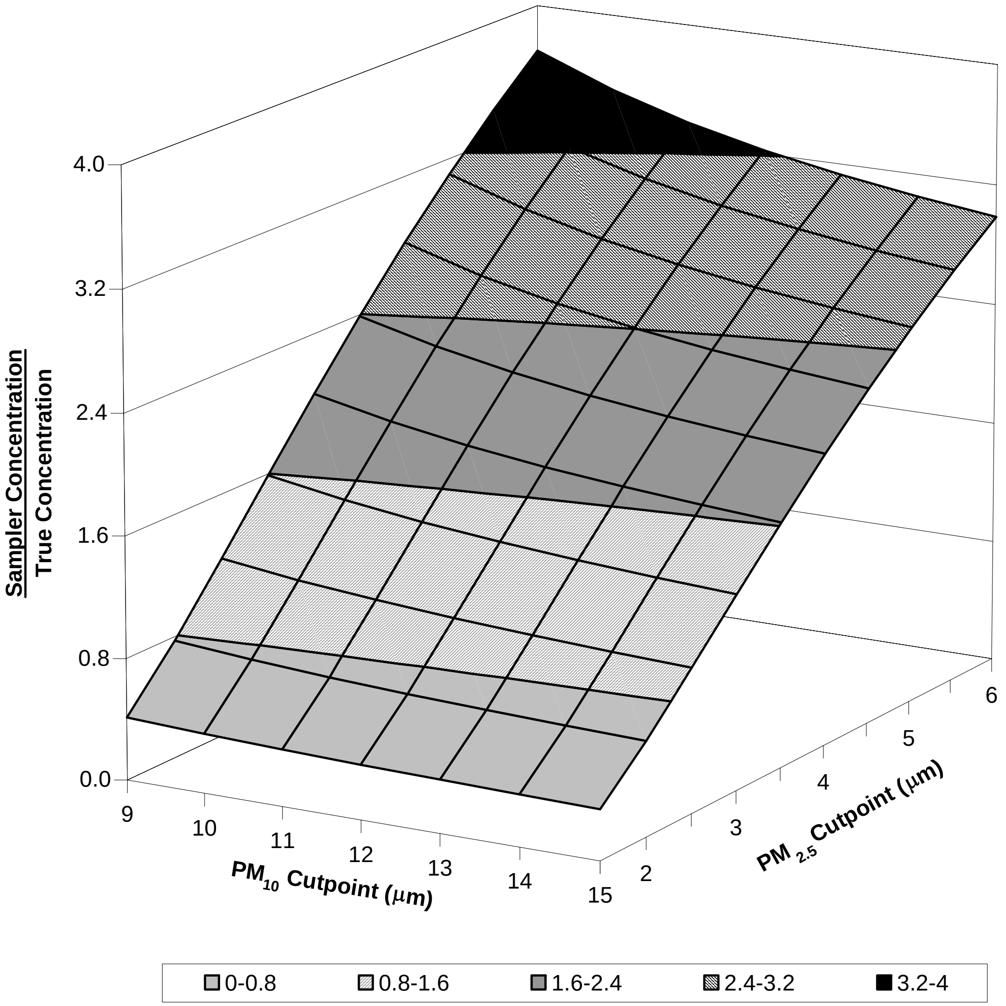


**Figure 59.** Theoretical ratios of PM$_{2.5/10}$ to true PSD concentrations for varying PM$_{2.5}$ and PM$_{10}$ sampler cutpoints (PSD – MMD = 5.7 μm and GSD = 2.25; PM$_{2.5}$ sampler slope = 1.3; PM$_{10}$ sampler slope = 1.5).

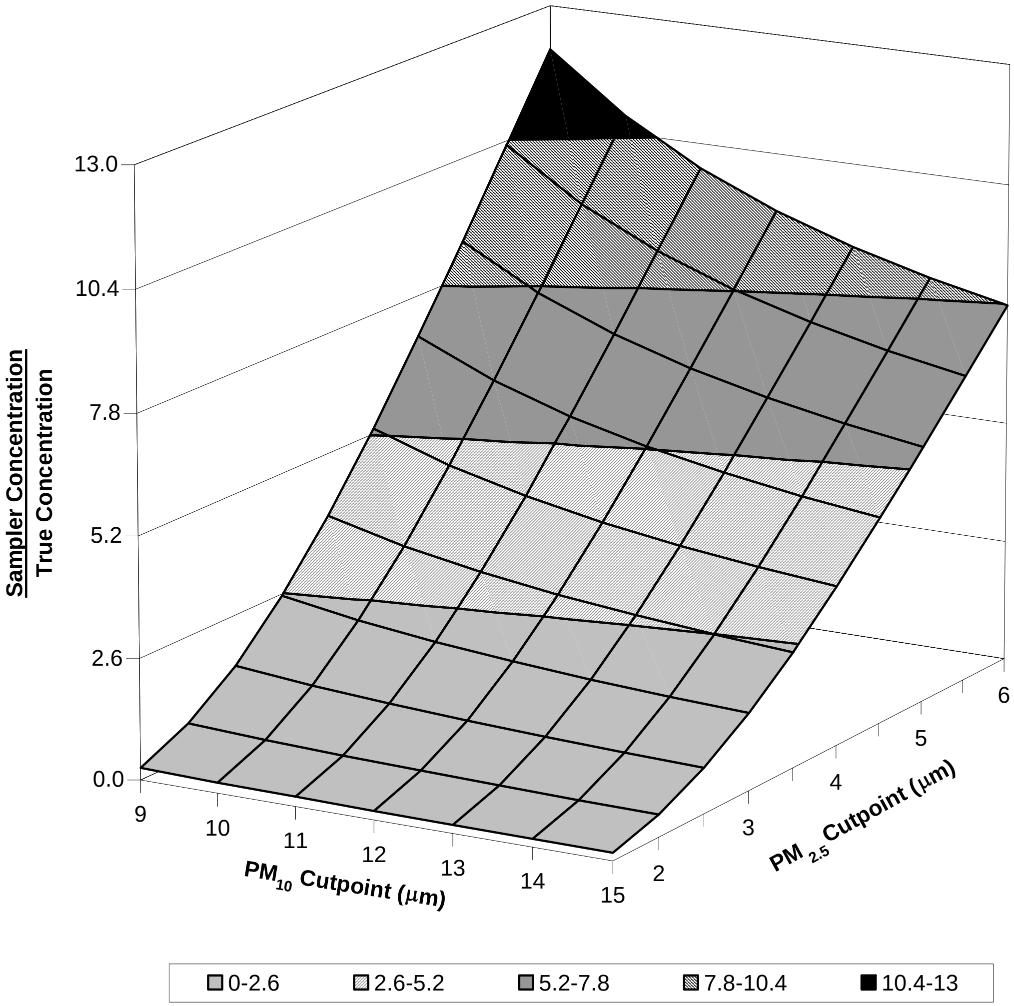


**Figure 60.** Theoretical ratios of $PM_{2.5/10}$ to true PSD concentrations for varying $PM_{2.5}$ and $PM_{10}$ sampler cutpoints (PSD – MMD = 10 μm and GSD = 2.0; $PM_{2.5}$ sampler slope = 1.3; $PM_{10}$ sampler slope = 1.5).

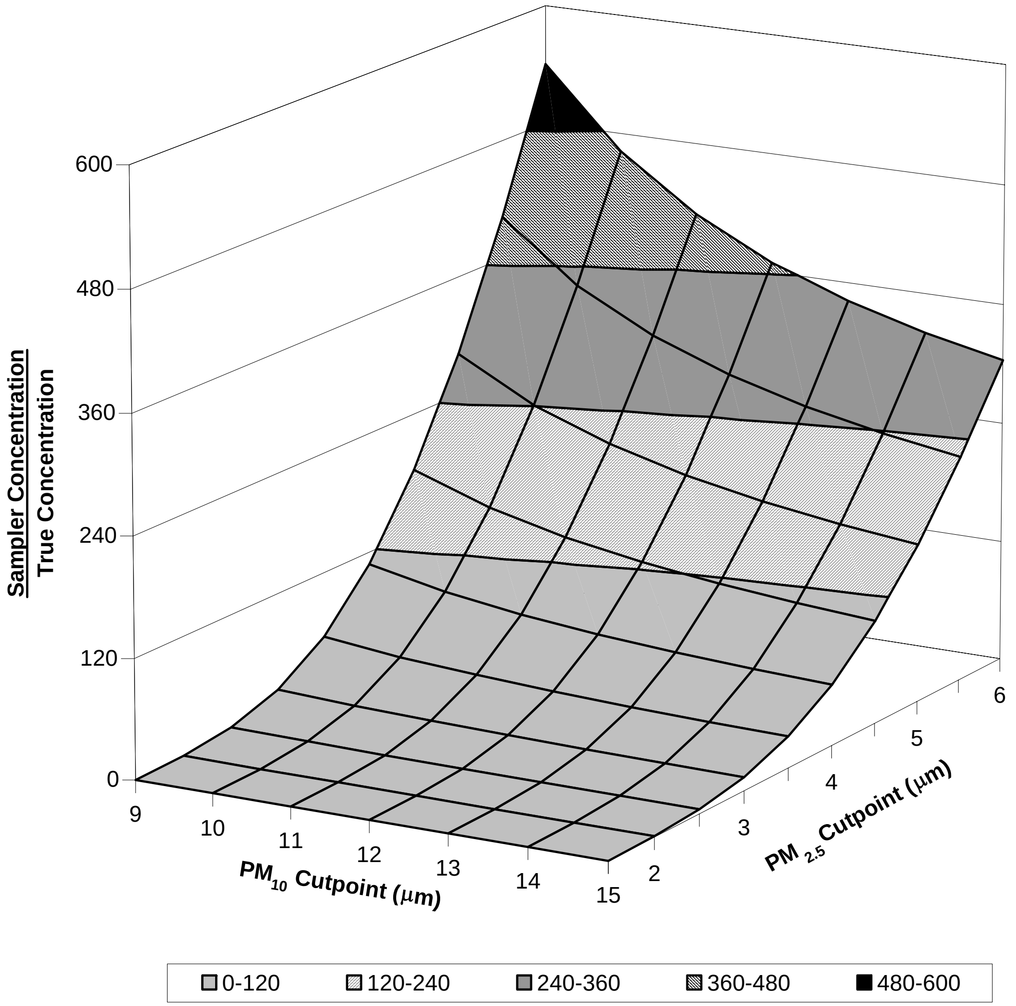


**Figure 61.** Theoretical ratios of $PM_{2.5/10}$ to true PSD concentrations for varying $PM_{2.5}$ and $PM_{10}$ sampler cutpoints (PSD – MMD = 10 μm and GSD = 1.5; $PM_{2.5}$ sampler slope = 1.3; $PM_{10}$ sampler slope = 1.5).

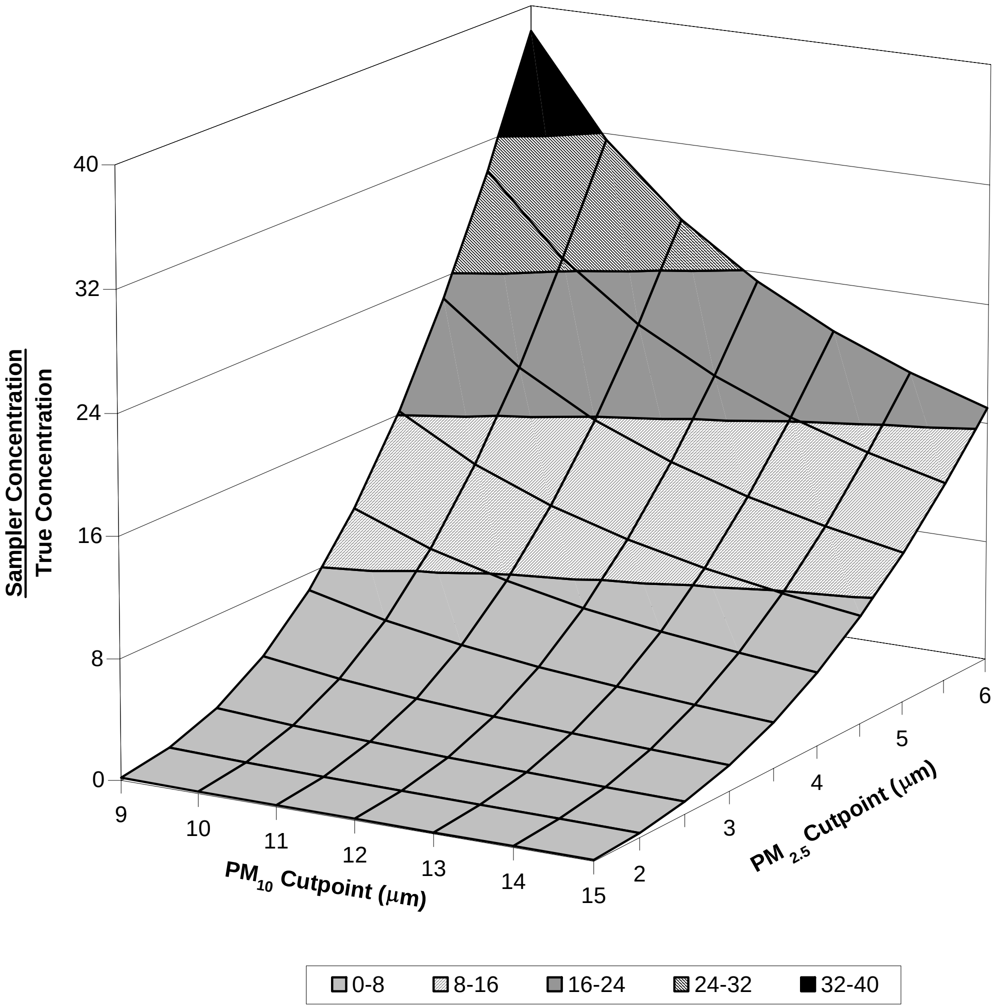


**Figure 62.** Theoretical ratios of $PM_{2.5/10}$ to true PSD concentrations for varying $PM_{2.5}$ and $PM_{10}$ sampler cutpoints (PSD – MMD = 20 μm and GSD = 2.0; $PM_{2.5}$ sampler slope = 1.3; $PM_{10}$ sampler slope = 1.5).

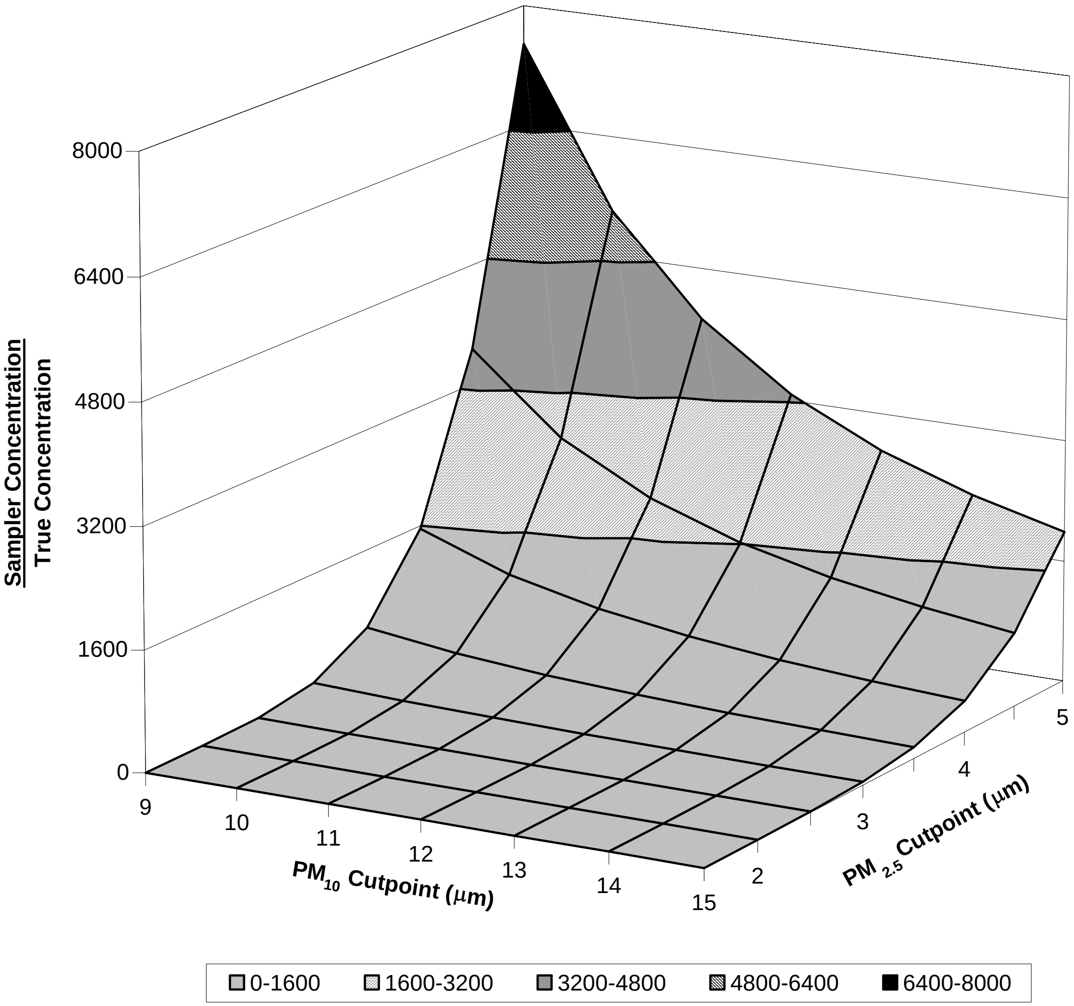


**Figure 63.** Theoretical ratios of $PM_{2.5/10}$ to true PSD concentrations for varying $PM_{2.5}$ and $PM_{10}$ sampler cutpoints (PSD – MMD = 20 μm and GSD = 1.5; $PM_{2.5}$ sampler slope = 1.3; $PM_{10}$ sampler slope = 1.5).

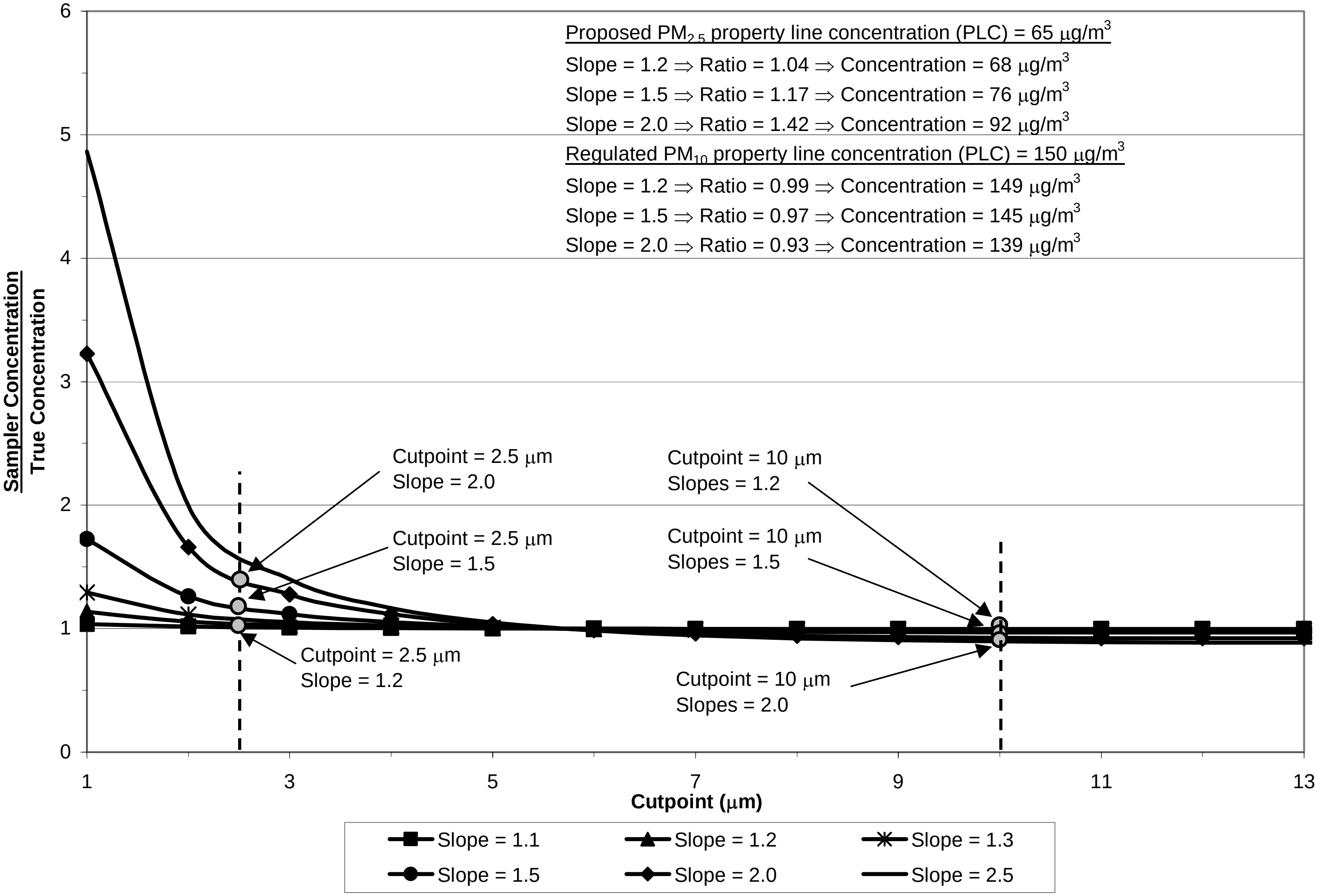


**Figure 64.** Effect of sampler cutpoint and slope on the sampler to true concentration ratio when exposed to a dust with a MMD of 5.7 μm and a GSD of 2.25.

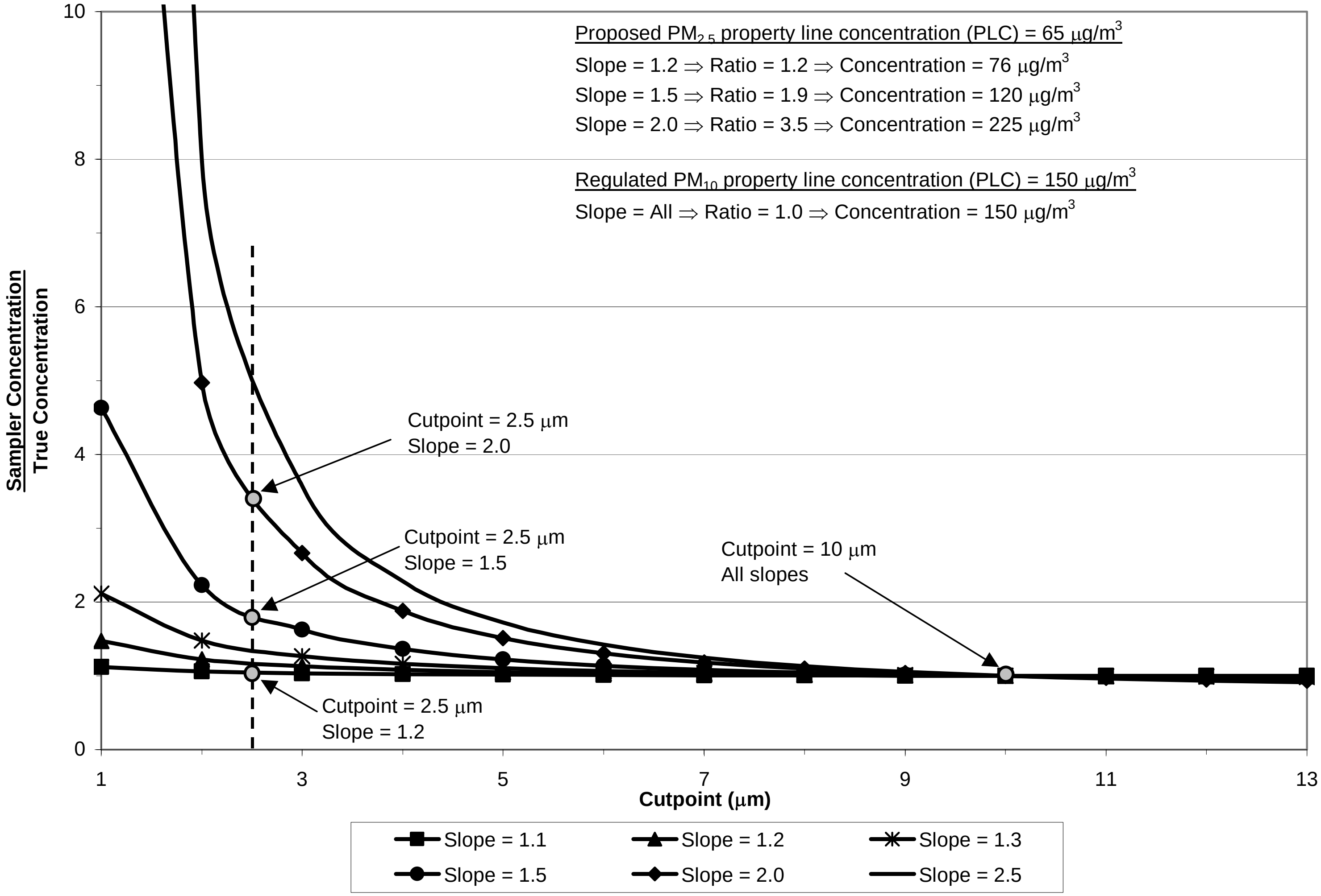


**Figure 65.** Effect of sampler cutpoint and slope on the sampler to true concentration ratio when exposed to a dust with a MMD of 10 μm and a GSD of 2.0.

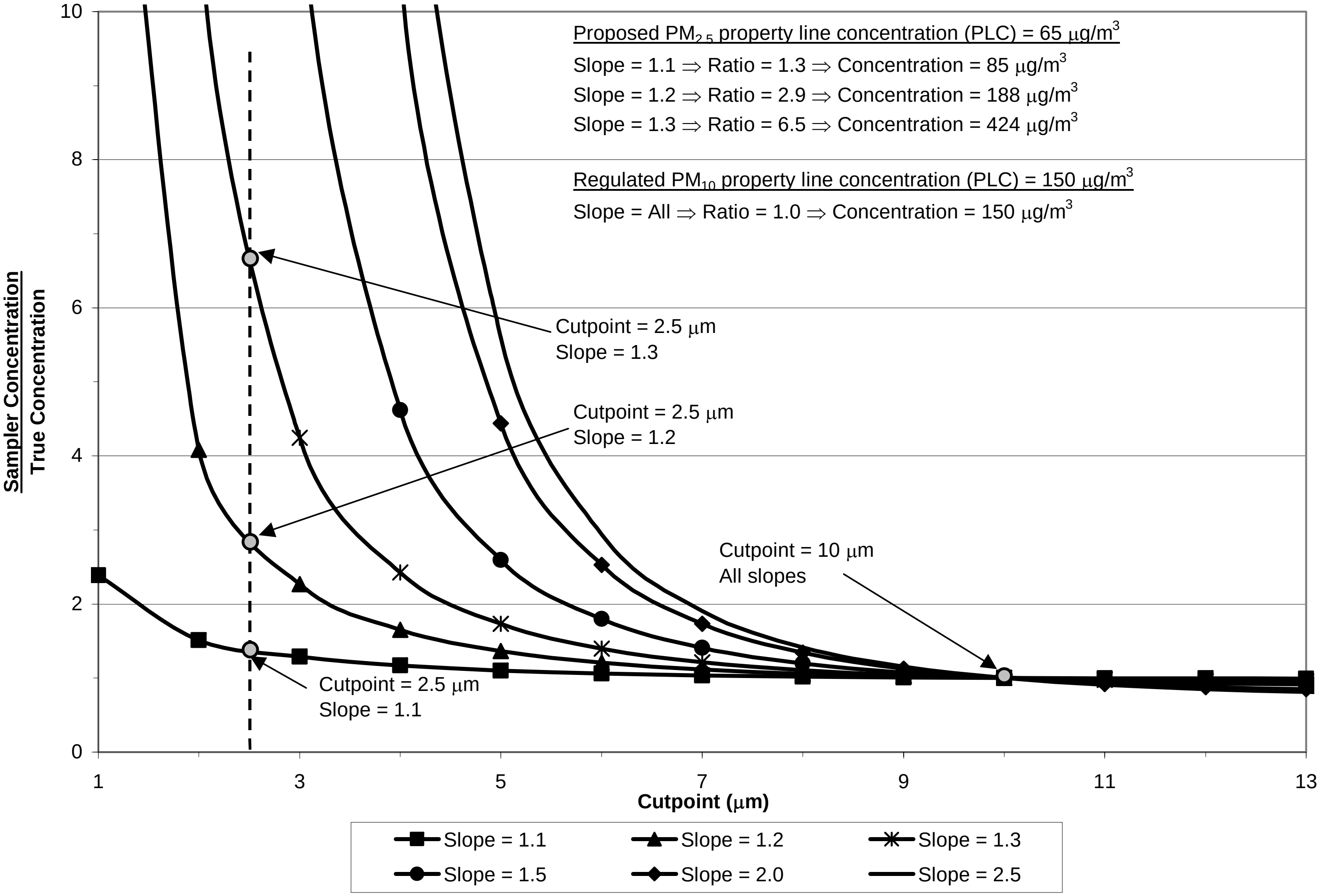


**Figure 66.** Effect of sampler cutpoint and slope on the sampler to true concentration ratio when exposed to a dust with a MMD of 10 μm and a GSD of 1.5.

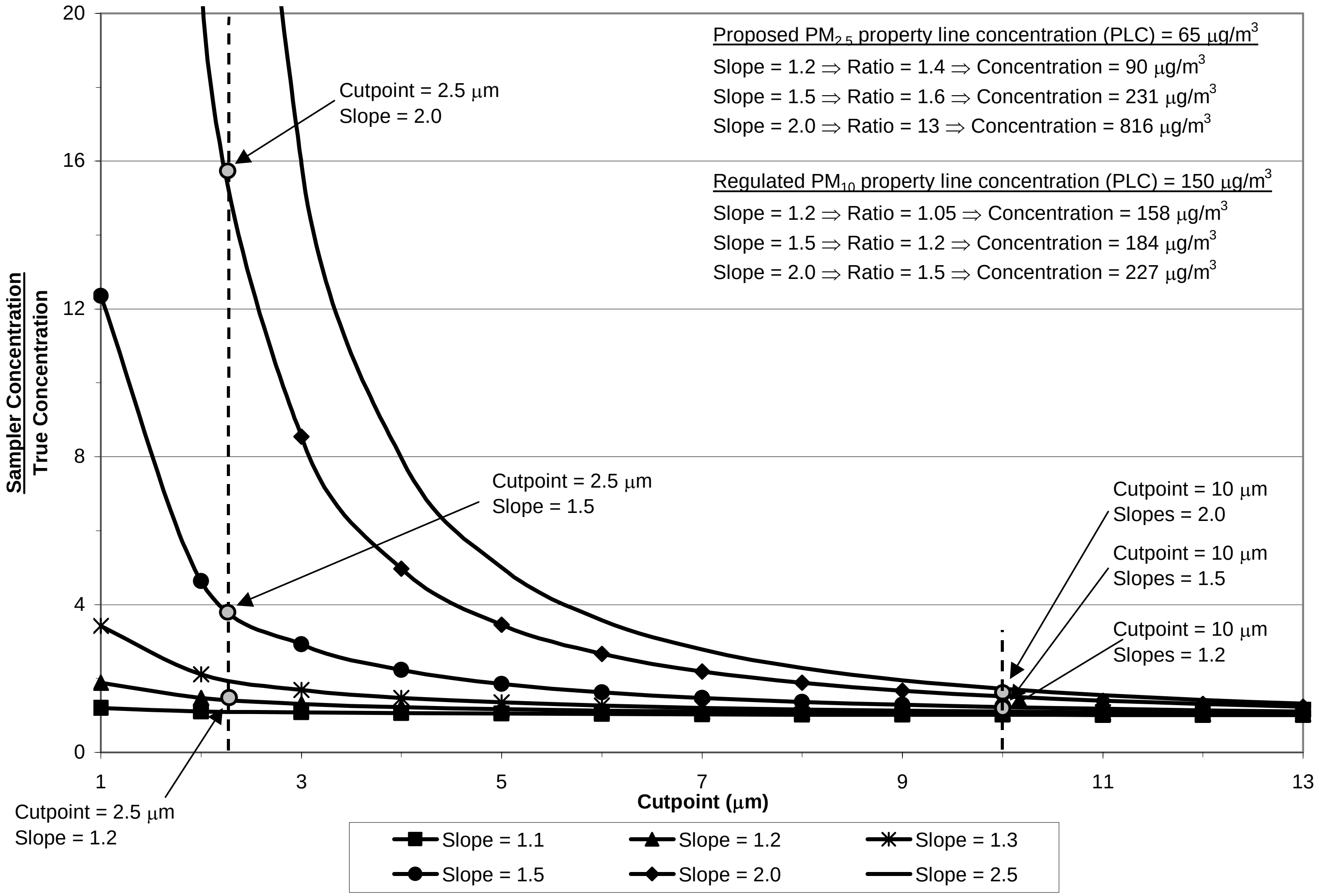


**Figure 67.** Effect of sampler cutpoint and slope on the sampler to true concentration ratio when exposed to a dust with a MMD of 20 μm and a GSD of 2.0.

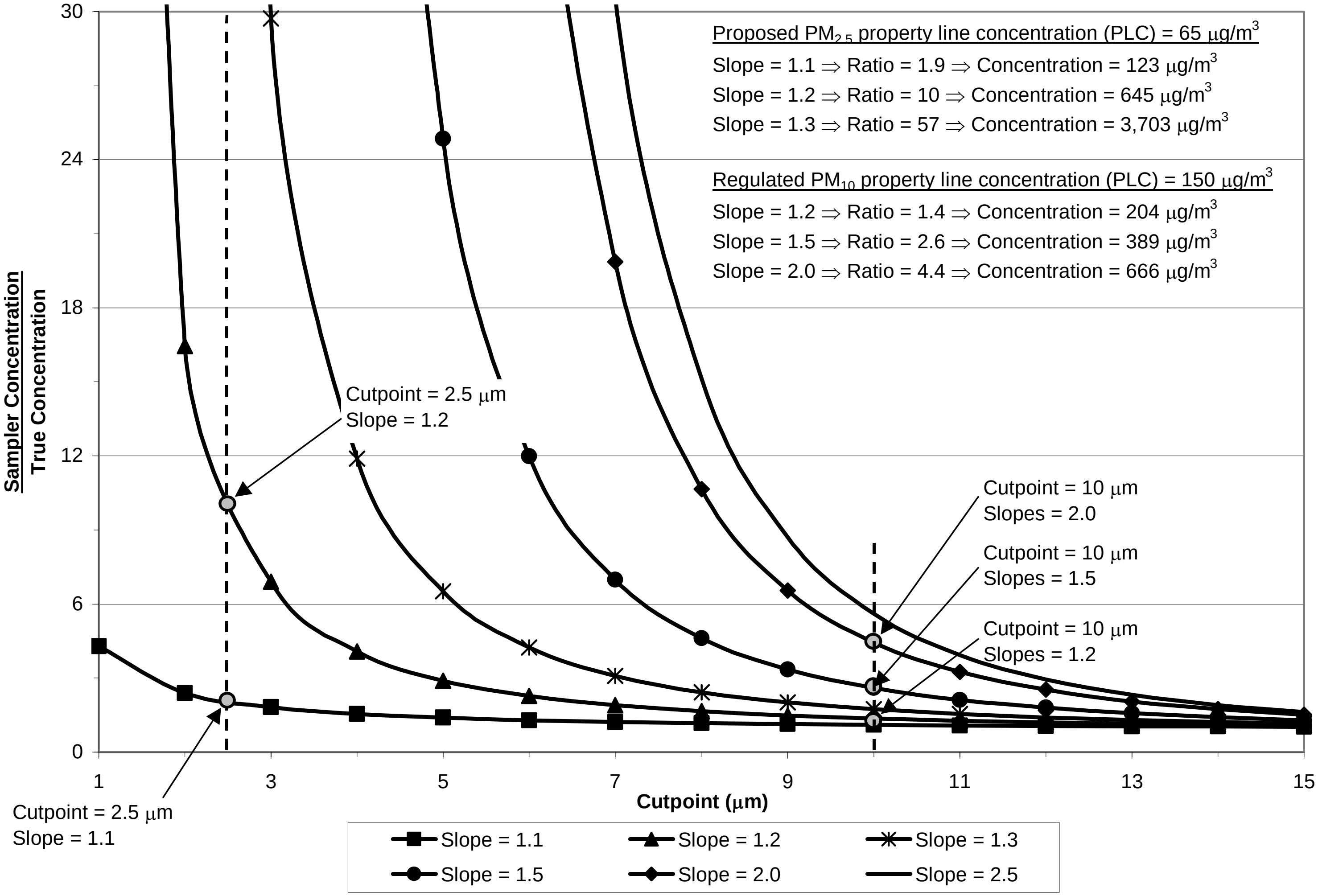


**Figure 68.** Effect of sampler cutpoint and slope on the sampler to true concentration ratio when exposed to a dust with a MMD of 20 μm and a GSD of 1.5.

## Cotton Gin PSDs

The results of the cotton gin PSD analyses are divided into five sections: 1) Stoneville gin trash PSDs; 2) Lubbock gin trash PSDs; 3) Idria Gin #1 emission factors and corresponding PSDs for the gins 1st pre-cleaning system; 4) Mesa Farmers Cooperative Gin emission factors and corresponding PSDs for the various exhausts tested by EEMC; and 5) estimating PSD characteristics based on the EPA 1996 AP-42 list of cotton gin emission factors.

### Stoneville – Gin Trash PSD Analysis

All samples received were air-washed and a particle size analysis was completed on the poly-web filters used in the air-wash procedure. There were a total of ten treatments and three replications with three Coulter repeated measures. The average PSDs for each treatment are provided in Appendix E. The PROC Mixed (Littell et al., 1996) procedure in SAS was utilized to compare the PSD characteristics associated with each of the treatments. The variability associated with the lint cleaner and gin stand treatments was excessive; therefore, the lint cleaner gin stand treatments were removed from the statistical analysis.

Significant difference were detected ($\alpha = 0.05$) for the machine, harvest (picker or stripper), and the interaction of machine and harvest. Results of the particles size analysis are reported in Table 20. The effect attributed to harvesting method was only significant in determining the MMD of the PSD. The MMD for all treatments ranged from 16.1 to 21.4 μm and the GSD ranged from 2.1 to 2.3 for all treatments. The percent of particles less than 10 μm ranged from 15.4 to 28.5% and the percent of particles less than 2.5 μm ranged from 0.20 to 1.25% for all treatments. The mean square error (MSE) associated with the percent of particles less than 10 μm was relatively high, indicating a large within treatment variability. No significant differences were detected in the particle density measurements. An average particle density of 1.95 g/cm$^3$ was determined.

**Table 20.** Average particle size distribution characteristics determined by Coulter Counter analysis for material less than 100 μm captured from individual machines of the Stoneville, MS, USDA-ARS Cotton Ginning Research Unit's Micro-Gin.

| Harvest Method | Machine | Coulter Counter PSD Characteristics | | | |
|---|---|---|---|---|---|
| | | MMD (μm)* | GSD | % $PM_{10}$ | % $PM_{2.5}$ |
| Picker | | | | | |
| | 1st Cylinder Cleaner | 18.1 c | 2.1 d | 20.7 bc | 0.33 bc |
| | 2nd Cylinder Cleaner | 18.1 c | 2.2 b | 22.7 b | 0.60 b |
| | 1st Stick Machine | 19.5 bc | 2.1 c | 18.7 cd | 0.31 bc |
| Stripper | | | | | |
| | 1st Cylinder Cleaner | 20.7 ab | 2.1 cd | 16.3 d | 0.21 bc |
| | 2nd Cylinder Cleaner | 21.4 a | 2.1 cd | 15.4 d | 0.20 c |
| | 1st Stick Machine | 16.3 d | 2.2 ab | 26.9 a | 1.00 a |
| | 2nd Stick Machine | 16.1 d | 2.3 a | 28.5 a | 1.25 a |
| MSE | | 2.05 | 0.004 | 13.41 | 0.1208 |
| p-value | | | | | |
| | Machine | < 0.0001 | < 0.0001 | < 0.0001 | < 0.0001 |
| | Harvest | 0.0043 | 0.2774 | 0.1447 | 0.4439 |
| | Machine*Harvest | < 0.0001 | < 0.0001 | < 0.0001 | < 0.0001 |
| F-value | | | | | |
| | Machine | 28.98 | 27.46 | 29.25 | 23.57 |
| | Harvest | 8.55 | 1.19 | 2.16 | 0.59 |
| | Machine*Harvest | 46.26 | 18.45 | 37.91 | 19.78 |

* MMD's reported in terms of aerodynamic equivalent diameter with a particle density of 1.96 g/$cm^3$.

Means in the same column followed by the same letter are not statistically different at the 0.05 level of significance.

### Lubbock – Gin Trash PSD Analysis

Eight of the twenty-four samples shipped from the Lubbock Laboratory were lost during shipping. These samples included: 1) variety HS200, field cleaned, unloading system; 2) variety HS200, field cleaned, feeder and gin stand; 3) variety HS200, field cleaned, lint cleaner; 4) variety HS26, field cleaned, unloading; 5) variety HS26, field cleaned, inclines; 6) variety HS26, field cleaned, extractors; 7) variety HS26, non-field cleaned, extractors; and 8) variety HS26, non-field cleaned, lint cleaners. All samples received were air-washed and a particle size analysis was completed on the poly-web filters used in the air-wash procedure.

There were a total of sixteen treatments and three replications with three Coulter repeated measures. The average PSDs for each treatment are provided in Appendix F. The PROC Mixed (Littell et al., 1996) procedure in SAS was utilized to compare the

PSD characteristics associated with each of the treatments. The PROC Mixed procedure was selected because of the procedures ability to deal with missing data. The variability associated with the lint cleaner treatments was excessive; therefore, the lint cleaner treatments were removed from the statistical analysis.

There were no cleaning method (field cleaned or non-field cleaned) or cleaning method interaction effects at a $\alpha$ of 0.05 for any of the PSD characteristic measurements. Further, there were no system by variety interactions at the 0.05 level. A summary of the particle size analysis results is provided in Table 21. Significant differences attributed to processing system and variety were detected for all particle size characteristics at the 0.05 $\alpha$ level. The MSE associated with the MMD and percent of particles smaller than 10 $\mu$m was relatively high, indicating a high within sample variation. In general, the MMDs ranged from 18.3 to 24.1 $\mu$m and the GSDs ranged from 1.9 to 2.1. The percent of PM less than 10 $\mu$m ranged from 10.0 to 21.6% and the percent of PM less than 2.5 $\mu$m ranged from 0.05 to 0.47%. When focusing on the variety effects, the MMD associated with the HS26 variety is larger than that for the HS200 variety, which resulted in lower $PM_{10}$ and $PM_{2.5}$ percentages for the HS26 variety as compared to the HS200 variety. When examining the PSD characteristics associated with the all gin systems combined treatment, the average MMDs were 20.0 to 24.1 $\mu$m with GSDs of 2.1 and 2.0 for the HS200 and HS26 varieties, respectively.

No significant variety, processing system, cleaning method (field cleaned or non-field cleaned) differences or interaction effects at a $\alpha$ of 0.05 were detected for the particle density measurements. The average particle density for all treatments was 1.97 g/cm$^3$. When comparing the particle densities associated with the Stoneville and Lubbock samples, no significant differences were detected at the 0.05 level. Further, the average particle density associated with the Lubbock and Stoneville samples combined was 1.96 g/cm$^3$. Therefore, a particle density of 1.96 g/cm3 was used in reporting all PSDs associated with the Lubbock, Stoneville, Idria Gin #1, and Mesa Farmers Cooperative Gin samples.

**Table 21.** Average particle size distribution characteristics determined by Coulter Counter analysis for material less than 100 μm captured from individual or sequences of machines of the Lubbock, TX, USDA-ARS Cotton Production and Processing Research Unit's commercial size gin.

| | | Mean | | | |
|---|---|---|---|---|---|
| Variety | System | MMD (μm)* | GSD | $PM_{10}$ (%) | $PM_{2.5}$ (%) |
| HS26 | | | | | |
| | Unloading | 19.9 ab | 1.9 bc | 14.9 abcd | 0.10 bc |
| | Incline Cleaners | 22.3 ab | 2.0 abc | 11.9 bcd | 0.09 bc |
| | Feeder and Gin Stand | 21.5 ab | 1.9 c | 12.0 bcd | 0.05 c |
| | Total | 24.1 a | 2.0 bc | 10.0 d | 0.05 c |
| HS200 | | | | | |
| | Unloading | 18.9 b | 2.0 abc | 17.8 abcd | 0.20 abc |
| | Incline Cleaners | 18.8 b | 2.1 ab | 20.0 ab | 0.34 ab |
| | Extractors | 18.3 b | 2.1 a | 21.6 a | 0.47 a |
| | Feeder and Gin Stand | 22.3 ab | 2.0 abc | 11.7 bcd | 0.06 bc |
| | Total | 20.0 ab | 2.1 ab | 17.5 abcd | 0.23 abc |
| MSE | | 4.45 | 0.0059 | 15.87 | 0.0202 |
| p-value | | | | | |
| | Machine | 0.0326 | 0.0091 | 0.0210 | 0.0028 |
| | Variety | 0.0105 | 0.0051 | 0.0021 | 0.0097 |
| | Machine*Variety | 0.0585 | 0.7908 | 0.0988 | 0.3501 |
| F-value | | | | | |
| | Machine | 2.99 | 4.03 | 3.34 | 5.03 |
| | Variety | 7.36 | 9.01 | 11.15 | 7.54 |
| | Machine*Variety | 2.75 | 0.35 | 2.27 | 1.13 |

* MMD's reported in terms of aerodynamic equivalent diameter with a particle density of 1.96 g/cm$^3$.

Means in the same column followed by the same letter are not statistically different at the 0.05 level of significance.

Idria Gin #1 – Emission Factor and PSD Analyses of the 1st Pre-Cleaning System

$AIR_x$ testing provided the parameter values that were used in equation 29 to calculate the sampler based emission factors. The parameters values provided by AIRx testing, the calculated sampler and true emission factors for each individual treatment, and the average PSDs for each of the treatments are provided in Appendix G. The stack area used in equation 29 was 0.34 m$^2$ (3.69 ft$^2$), for all calculations. Table 22 provides the summary of the particle size distribution analyses. The PROC Mixed (Littell, et al., 1996) procedure in SAS was utilized to compare the PSD characteristics associated with 1st and 2nd pick Pima cotton. The test data were categorized by three class variables; 1) harvest (1st or 2nd pick), 2) sample location (cyclone wash, post-cyclone wash, or filter), and 3) test number (replication). Test number was treated as a random effect, while harvest and sample location were treated as fixed effects.

**Table 22.** Particle size distribution characteristics as determined by Coulter Counter analyses for Idria Gin #1's 1st seed cotton cleaning and drying system exhaust.

| PSD Characteristic | Location | Mean<br>1st Pick | <br>2nd Pick | MSE | P-value | F-value |
|---|---|---|---|---|---|---|
| MMD (μm)* | | | | | | |
| | Filter | 8.4 a | 9.0 a | 1.344 | 0.1300 | 2.41 |
| | Post-Cyclone | 17.1 a | 15.6 b | 2.884 | 0.0147 | 6.61 |
| | Cyclone | 19.9 a | 18.8 a | 6.373 | 0.1855 | 1.83 |
| GSD | | | | | | |
| | Filter | 1.6 a | 1.6 a | 0.000 | 0.1042 | 2.79 |
| | Post-Cyclone | 1.9 a | 1.8 b | 0.003 | < 0.0001 | 44.52 |
| | Cyclone | 1.7 b | 1.8 a | 0.003 | < 0.0001 | 27.29 |
| $PM_{10}$ (%) | | | | | | |
| | Filter | 66.0 a | 60.6 a | 1.138 | 0.1393 | 2.29 |
| | Post-Cyclone | 21.0 a | 22.5 a | 0.268 | 0.3899 | 0.76 |
| | Cyclone | 10.0 a | 13.8 a | 0.404 | 0.0998 | 2.88 |
| $PM_{2.5}$ (%) | | | | | | |
| | Filter | 0.371 a | 0.309 a | 4.1 e-4 | 0.3666 | 0.84 |
| | Post-Cyclone | 0.213 a | 0.090 b | 2.2 e-4 | 0.0177 | 6.21 |
| | Cyclone | 0.014 a | 0.026 a | 7.8 e-6 | 0.2038 | 1.69 |

* MMD's reported in terms of aerodynamic equivalent diameter with a particle density of 1.96 g/cm$^3$.
Means in the same row followed by the same letter are not statistically different at the 0.05 level of significance.

There was no harvest by sample location interaction effects for any of the PSD characteristic measurements. The effect attributed to sample location was much more significant than the effect attributed to harvest for the MMD, percent $PM_{10}$, and percent $PM_{2.5}$. Therefore, the effects attributed to harvest were analyzed within sample location. As expected, the percent $PM_{10}$ and percent $PM_{2.5}$ were larger for the PM captured on the filters than the PM associated with the two washes. Generally, there were no significant effects ($\alpha = 0.05$) between 1st and 2nd pick Pima cotton for the PSD characteristics of any sample location; however, there were a few exceptions. There were significant differences in GSD values attributed to harvest for the cyclone and post-cyclone washes. Although differences were detected in the GSD values, the differences are relatively small and can most likely be neglected. The main significant differences detected between 1st and 2nd pick where MMD and percent $PM_{2.5}$ for the post-cyclone wash. Both the MMD and percent $PM_{2.5}$ associated with the post-cyclone wash were significantly smaller ($\alpha = 0.05$) for the 2nd pick Pima cotton as compared to the 1st pick.

Equation 29 was used to calculate the source sampling TSP and $PM_{10}$ emission factors using the data provided in Tables 35 and 36 of Appendix G. Further, the true $PM_{10}$ and $PM_{2.5}$ emission factors were calculated as previously described using the information provided in Tables 35 and 36 of Appendix G. Table 37 of Appendix G includes the source sampling emission factors for TSP and $PM_{10}$, the source sampling ratio of $PM_{10}$ to TSP, the true $PM_{10}$ and $PM_{2.5}$ emission factors, and the true ratio of $PM_{10}$ to TSP and $PM_{2.5}$ to TSP for the individual treatments. The average values corresponding to Table 37 of Appendix G are shown in Table 23. The PROC Mixed (Littell, et al., 1996) procedure in SAS was utilized to compare the emission factors and emission factor ratios associated with 1st and 2nd pick Pima cotton.

**Table 23.** Idria Gin #1's 1st seed cotton cleaning and drying system's exhaust emission factors as determined by source sampling and Coulter Counter analysis.

| | Source Sampling | | | Coulter Counter Analysis (True) | | | |
|---|---|---|---|---|---|---|---|
| Harvest | TSP** | $PM_{10}$** | $PM_{10}$/TSP Ratio | $PM_{10}$** | $PM_{2.5}$** | $PM_{10}$/TSP Ratio | $PM_{2.5}$/TSP Ratio |
| 1st Pick | 0.0857 | 0.0541 | 63.1 % | 0.0350 | $1.9\ e^{-4}$ | 40.8 % | 0.22 % |
| | (0.1889) | (0.1192) | | (0.0771) | $(4.3\ e^{-4})$ | | |
| 2nd Pick | 0.0723 | 0.0407 | 56.1 % | 0.0264 | $1.2\ e^{-4}$ | 35.9 % | 0.16 % |
| | (0.1595) | (0.0898) | | (0.0582) | $(2.7\ e^{-4})$ | | |
| | | | | | | | |
| MSE | 0.0006 | 0.0003 | 0.1313 | 0.0003 | $5.3\ e^{-8}$ | 0.5083 | $1.4\ e^{-}6$ |
| p-value*** | 0.0668 | 0.0108 | 0.0076 | 0.0977 | 0.2645 | 0.2615 | 0.3621 |
| F-value | 4.23 | 9.77 | 11.10 | 3.34 | 1.40 | 1.42 | 0.91 |

** TSP, $PM_{10}$, and $PM_{2.5}$ emission factors reported in kg of PM emitted per bale of lint cotton produced (lb of PM emitted per bale of lint cotton produced).

*** p-value is the test of significance of means being equal in the PROC Mixed procedure of SAS. Any p-value greater than 0.05 was considered to be non-significant.

Emission factors correspond to only the one cyclone tested; these emission factors must be multiplied by 4 to determine the total emission factor for the #1 pre-cleaning system.

There were no significant differences ($\alpha = 0.05$) in the processing rates between 1st and 2nd pick Pima cotton (16.60 and 16.36 bales/h respectively). The $PM_{10}$ source sampling emission factors and corresponding $PM_{10}$ to TSP ratios (Table 23) were significantly higher for 1st pick Pima cotton as compared to 2nd pick ( 0.2163 to 0.1629 kg/bale and 63 to 56%, respectively). No significant differences ($\alpha = 0.05$) were detected between 1st and 2nd pick Pima cotton TSP, true $PM_{10}$, or true $PM_{2.5}$ emission factors and no differences were detected between the true $PM_{10}$ or $PM_{2.5}$ to TSP ratios. The data does however show a trend of lower emission factors and smaller emission factor ratios for 2nd pick Pima cotton as compared to 1st pick.

The $PM_{10}$ emission factors and $PM_{10}$ to TSP ratios determined by source sampling and particle size analyses were compared to determine if these test data followed the theoretical errors associated with the current federal guidelines of measuring $PM_{10}$ via source sampling. The 1st pre-cleaning system $PM_{10}$ emission factors determined by source sampling, as given in Table 23, were significantly higher than the emission factors determined by particle size analyses for both 1st and 2nd pick Pima cotton (0.2163 to 0.1399 kg/bale and 0.1629 to 0.1056 kg/bale, respectively). The $PM_{10}$ to TSP ratios determined by source sampling were also significantly higher than the ratios determined by particle size analyses for both 1st and 2nd pick Pima cotton (63.1 to 40.8% and 56.1 to 35.9%, respectively). The ratio of the $PM_{10}$ emission factors determined by source sampling to the $PM_{10}$ emission factors determined by particle size analyses was 1.55. This implies that the $PM_{10}$ emission factors for the Idria Gin #1's 1st pre-cleaning system exhaust determined by EPA's Method 201A source sampling procedures were overestimated by 155%.

The TSP and $PM_{10}$ emission factors for the No. 1 dryer and cleaner listed in the 1996 AP-42 were 0.16 and 0.05 kg/bale (0.36 and 0.12 lb/bale), respectively. This AP-42 designation corresponds to the Idria Gin's 1st pre-cleaning system. The average Idria Gin #1 pre-cleaning system TSP and $PM_{10}$ source sampling emission factors for 1st pick Pima cotton were 0.35 and 0.22 kg/bale (0.76 and 0.48 lb/bale), respectively. The equivalent TSP and $PM_{10}$ emission factors for 2nd pick Pima cotton were 0.29 and 0.16

kg/bale (0.64 and 0.36 lb/bale), respectively. The $PM_{10}$ and $PM_{2.5}$ emission factors determined by particle size analyses were 0.14 and 0.0009 kg/bale (0.31 and 0.002 lb/bale) respectively for 1st pick and 0.01 and 0.0005 kg/bale (0.23 and 0.001 lb/bale) respectively for 2nd pick Pima cotton.

## Mesa Farmers Cooperative Gin – Emission Factor and PSD Analyses on Select Systems

EEMC testing provided the parameter values that were used in equation 29 to calculate the sampler based emission factors and provided the filter and wash samples for the particle size analysis. Average PSDs for the TSP filters, TSP wash, $PM_{10}$ filter, and $PM_{10}$ washes for each exhaust tested are provided in Appendix H. The TSP values used in equation 29 to calculate the TSP, true $PM_{10}$, and true $PM_{2.5}$, based on Method 5 sampling are provided in Tables 38 and 42 of Appendix H. The $PM_{10}$ values used in equation 29 to calculate the TSP, true $PM_{10}$, and true $PM_{2.5}$, based on Method 201a sampling are provided in Tables 39 and 41 of Appendix H. The sampling time used in equation 29 for all emission factor calculations was 60 minutes.

One important issue associated with the Method 201a sampling data that should be noted is the highly sporadic isokinetic variation values. These values ranged from 85.6 to 130.2%. According to EPA Method 201a guidelines, the tolerances for Method 201A isokinetic variation is 80% to 120%. When isokinetic variation is above 100%, emission factors are under-estimated. However, conducting $PM_{10}$ sampling with an isokinetic variation above 100% will bias the results towards smaller particle sizes. In other words, when the isokinetic variation is above 100% the sampling system is pulling air at a higher velocity than the corresponding stack air velocity, which will cause smaller particles, outside of the normal sampling streamlines, to enter the sampling system while the larger particles will deviate from the streamlines and bypass the sampling system.

The average particle size characteristics associated with the TSP filters, TSP wash, $PM_{10}$ filter, $PM_{10}$ cyclone wash, and the $PM_{10}$ post-cyclone wash are provided in Table 24. The individual replication data for the TSP and $PM_{10}$ sampling are provided in Table 40 of Appendix H. There were significant differences for all PSD characteristics. In general, the MMD associated with the washes were larger than the MMDs associated with the filters, for both Method 5 and Method 201a sampling. Further, the fractions of $PM_{10}$ and $PM_{2.5}$ were larger for the PM captured on the filters than the PM contained in the wash.

The average TSP, true $PM_{10}$, and true $PM_{2.5}$ emission factors, based on Method 5 sampling, are provided in Table 25. No significant differences, at $\alpha$ equal 0.05, were detected for the ratio of true $PM_{2.5}$ to TSP. However, significant differences attributed to process system exhaust were detected for the ratio of true $PM_{10}$ to TSP and the TSP and $PM_{10}$ emission factors. In general, the unloading system had the highest emission factors and the third incline cleaner exhaust, system A (#3A pull), had the lowest emission factors.

The average TSP, $PM_{10}$, true $PM_{10}$, and true $PM_{2.5}$ emission factors, based on Method 201a, are provided in Table 26. No significant process system exhaust differences were detected for the ratio of $PM_{10}$ to TSP or the ratio of true $PM_{2.5}$ to TSP. However, significant differences were detected for all emission factors and the ratio of true $PM_{10}$ to TSP. The unloading system had the highest TSP and $PM_{10}$ emission factors, while the third incline cleaner exhaust, system A (#3A pull), had the lowest emission factors. Although TSP emission factors determined from Method 201a are reported, these emission factors should not be used in place of the TSP emission factors determined by Method 5, because of the differences associated with the sampling procedures.

**Table 24.** Average particle size distribution characteristics determined by Coulter Counter analysis for the Mesa Gin exhausts sampled by EEMC in December of 2001.

| PSD | | Method 201a Sampling | | | Method 5 Sampling | |
|---|---|---|---|---|---|---|
| Characteristic | Location | Filter | Post-Cyclone | Cyclone | Filter | Wash |
| MMD (μm)* | | | | | | |
| | Unloading | 6.6 c | 8.9 b | 10.8 a | 9.4 b | 8.4 b |
| | # 1 Pull | 6.5 c | 15.0 a | 21.2 b | 7.1 d | 17.4 a |
| | # 2 Pull | 7.0 b | 8.2 b | 13.8 a | 7.8 c | 12.7 ab |
| | #3A Pull | 7.3 b | 10.9 b | 12.7 a | 7.4 cd | 8.7 b |
| | #3B Pull | 8.2 a | 10.5 b | 14.2 a | 10.8 a | 10.0 b |
| | Lint Basket Pull | 8.2 a | 11.2 ab | 14.9 a | 10.8 a | 16.2 a |
| | MSE | 0.083 | 7.948 | 14.633 | 0.164 | 12.900 |
| | p-value | < 0.0001 | 0.0001 | < 0.0001 | < 0.0001 | < 0.0001 |
| | F-value | 61.70 | 6.50 | 7.54 | 156.84 | 10.44 |
| GSD | | | | | | |
| | Unloading | 1.6 b | 1.8 ab | 1.8 c | 2.0 a | 1.8 b |
| | # 1 Pull | 1.5 b | 1.6 b | 2.1 a | 1.7 c | 2.1 a |
| | # 2 Pull | 1.6 b | 1.6 b | 1.9 b | 1.8 b | 1.7 b |
| | #3A Pull | 1.6 b | 2.0 a | 1.9 b | 1.8 b | 1.8 b |
| | #3B Pull | 1.8 a | 1.8 ab | 1.9 b | 2.1 a | 1.8 b |
| | Lint Basket Pull | 1.8 a | 2.0 a | 1.9 bc | 2.1 a | 1.8 b |
| | MSE | 0.003 | 0.017 | 0.009 | 0.009 | 0.022 |
| | p-value | < 0.0001 | < 0.0001 | < 0.0001 | < 0.0001 | 0.0004 |
| | F-value | 40.00 | 10.35 | 10.36 | 34.76 | 5.61 |
| PM$_{10}$ (%) | | | | | | |
| | Unloading | 83.4 ab | 61.9 a | 45.5 a | 53.4 d | 63.4 a |
| | # 1 Pull | 85.2 a | 31.3 b | 15.1 b | 75.1 a | 25.8 bc |
| | # 2 Pull | 79.0 bc | 65.6 a | 34.9 ab | 65.7 c | 44.5 ab |
| | #3A Pull | 74.9 c | 46.8 ab | 37.0 ab | 70.2 b | 60.0 a |
| | #3B Pull | 63.3 d | 48.5 ab | 32.7 ab | 46.0 e | 50.4 a |
| | Lint Basket Pull | 64.4 d | 45.1 ab | 32.8 ab | 45.5 e | 23.0 c |
| | MSE | 11.146 | 266.91 | 225.18 | 6.143 | 194.81 |
| | p-value | < 0.0001 | 0.0007 | 0.0050 | < 0.0001 | < 0.0001 |
| | F-value | 70.89 | 5.19 | 3.91 | 237.60 | 13.31 |
| PM$_{2.5}$ (%) | | | | | | |
| | Unloading | 1.33 bc | 1.72 a | 0.66 a | 2.8 ab | 2.2 a |
| | # 1 Pull | 1.08 c | 0.19 b | 0.19 a | 1.9 b | 0.4 c |
| | # 2 Pull | 1.05 c | 0.81 ab | 0.87 a | 3.1 ab | 0.7 c |
| | #3A Pull | 1.02 c | 1.40 a | 0.78 a | 3.2 a | 1.9 ab |
| | #3B Pull | 2.29 a | 0.89 ab | 1.04 a | 2.6 ab | 0.8 bc |
| | Lint Basket Pull | 1.85 ab | 1.63 a | 0.88 a | 2.6 ab | 0.3 c |
| | MSE | 0.1895 | 0.696 | 0.458 | 0.834 | 0.636 |
| | p-value | < 0.0001 | 0.0022 | 0.1475 | 0.0432 | < 0.0001 |
| | F-value | 12.89 | 4.42 | 1.73 | 2.50 | 9.03 |

* MMD's reported in terms of aerodynamic equivalent diameter with a particle density of 1.96 g/cm$^3$.

Means in the same column for the same PSD characteristic followed by the same letter are not statistically different at the 0.05 level of significance.

**Table 25.** Mesa Gin average exhaust emission factors as determined by Method 5 source sampling and Coulter Counter particle size analysis.

| | Source Sampling | | Coulter Counter Analysis (True) | | |
|---|---|---|---|---|---|
| Exhaust | TSP$^{**}$ | $PM_{10}$$^{**}$ | $PM_{2.5}$$^{**}$ | $PM_{10}$/TSP Ratio | $PM_{2.5}$/TSP Ratio |
| Unloading | 0.1417 | 0.0830 | 0.0037 | 57.3 a | 2.34 a |
| | (0.3124) a | (0.1829) a | (0.0081) a | | |
| # 1 Pull | 0.1233 | 0.0766 | 0.0018 | 62.1 a | 1.48 a |
| | (0.2717) b | (0.1688) b | (0.0040) b | | |
| # 2 Pull | 0.0234 | 0.0140 | 0.0005 | 58.0 a | 2.30 a |
| | (0.0515) a | (0.0308) a | (0.0012) ab | | |
| #3A Pull | 0.0042 | 0.0027 | 0.0001 | 64.3 a | 2.44 a |
| | (0.0093) b | (0.0059) b | (0.0002) b | | |
| #3B Pull | 0.0169 | 0.0080 | 0.0003 | 47.1 ab | 1.79 a |
| | (0.0373) b | (0.0175) b | (0.0007) b | | |
| Lint Basket Pull | 0.0246 | 0.0097 | 0.0005 | 39.7 b | 2.00a |
| | (0.0542) b | (0.0214) b | (0.0011) b | | |
| MSE | 0.0049 | 0.0019 | 0.0005 | 0.004 | 0.00005 |
| p-value$^{***}$ | 0.0004 | 0.0005 | 0.0099 | 0.0033 | 0.5506 |
| F-value | 10.78 | 10.38 | 5.08 | 6.75 | 0.83 |

* Emission factors correspond only to the one cyclone tested and not the entire system.

** TSP, $PM_{10}$, and $PM_{2.5}$ emission factors reported in kg of PM emitted per bale of lint cotton produced (lb of PM emitted per bale of lint cotton produced).

*** p-value is the test of significance of means being equal in the PROC Mixed procedure of SAS.

Means in the same column followed by the same letter are not statistically different at the 0.05 level of significance.

The true $PM_{10}$ emission factors determined from the Method 5 sampling are substantially lower than the $PM_{10}$ emission factors determined from Method 201A sampling. For the unloading, #1 pull, #2 pull, #3A pull, #3B pull, and lint basket pull the true $PM_{10}$ emission factors, determined by Method 5 sampling, are 16.5%, 13.3%, 26.4%, 48.2%, 28.8%, and 18.6% lower than the $PM_{10}$ emission factor determined by Method 201a sampling. Part of this difference could be attributed to the isokinetic variations previously discussed. When comparing the true $PM_{10}$ emission factors from the Method 5 sampling to the 1996 AP-42, the $PM_{10}$ emission factors from the unloading and 1st pre-cleaning systems are higher than the corresponding AP-42 emission factors. The true $PM_{10}$ emission factors from the Method 5 sampling are lower for the 2nd pre-cleaning, 3rd pre-cleaning, and lint basket pull systems than the corresponding AP-42 emission factors.

**Table 26.** Mesa Gin exhaust average emission factors as determined by Method 201a source sampling and Coulter Counter particle size analysis.

| | Source Sampling | | | Coulter Counter Analysis (True) | | | |
|---|---|---|---|---|---|---|---|
| Exhaust | TSP** | $PM_{10}$** | $PM_{10}$/TSP Ratio | $PM_{10}$** | $PM_{2.5}$** | $PM_{10}$/TSP Ratio | $PM_{2.5}$/TSP Ratio |
| Unloading | 0.1536<br>(0.3387) a | 0.0967<br>(0.2131) a | 65.4 a | 0.1090<br>(0.2403) a | 0.0018<br>(0.0041) a | 71.5 a | 1.23 a |
| # 1 Pull | 0.1323<br>(0.2917) a | 0.0868<br>(0.1913) a | 66.0 a | 0.0784<br>(0.1729) b | 0.0010<br>(0.0022) b | 59.6 ab | 0.74 a |
| # 2 Pull | 0.0240<br>(0.0529) b | 0.0177<br>(0.0389) b | 73.6 a | 0.0159<br>(0.0350) c | 0.0002<br>(0.0005) c | 66.3 ab | 0.98 a |
| #3A Pull | 0.0058<br>(0.0128) b | 0.0040<br>(0.0088) b | 70.6 a | 0.0033<br>(0.0072) c | 0.0001<br>(0.0001) c | 55.9 ab | 1.11 a |
| #3B Pull | 0.0194<br>(0.0427) b | 0.0103<br>(0.0227) b | 53.2 a | 0.0093<br>(0.0206) c | 0.0003<br>(0.0006) c | 48.6 ab | 1.46 a |
| Lint Basket Pull | 0.0253<br>(0.0558) b | 0.0115<br>(0.0253) b | 54.1 a | 0.0098<br>(0.0216) c | 0.0003<br>(0.0007) c | 42.1 b | 1.14 a |
| MSE | 0.0011 | 0.0001 | 0.0346 | 0.0003 | 1.59 e-7 | 0.0098 | 0.00002 |
| p-value*** | < 0.0001 | < 0.0001 | 0.6878 | < 0.0001 | < 0.0001 | 0.0304 | 0.5909 |
| F-value | 58.78 | 187.01 | 0.62 | 109.96 | 42.37 | 3.66 | 0.77 |

Emission factors correspond only to the one cyclone tested and not the entire system.

** TSP, $PM_{10}$, and $PM_{2.5}$ emission factors reported in kg of PM emitted per bale of lint cotton produced (lb of PM emitted per bale of lint cotton produced).

*** p-value is the test of significance of means being equal in the PROC Mixed procedure of SAS.

Means in the same column followed by the same letter are not statistically different at the 0.05 level of significance.

## Estimating PSD Characteristics Based on EPA's 1996 AP-42 List of Emission Factors

The $PM_{10}$ stack sampler characteristics used in this trial and error procedure were a $d_{50}$ of 11 μm and a slope of 1.76, as determined in the $PM_{10}$ stack sampler errors attributed to established tolerances section. In performing the trial and error calculations to determine the true percent $PM_{10}$ values, an anomaly was encountered with the lint cleaner emission factor. According to the 1996 AP-42, the TSP emission factor for lint cleaners with covered condenser drums is 0.5 kg/bale (1.1 lb/bale), while the $PM_{10}$ emission factor is assumed to be 50% of the TSP value. On the other hand, the TSP value for the lint cleaners with high-efficiency cyclones was 0.26 kg/bale (0.58 lb/bale), while the $PM_{10}$ emission factor was 0.11 kg/bale (0.24 lb/bale), resulting in a percent $PM_{10}$ of 41.4%. The problem associated with the two sets of emission factors is that the percent $PM_{10}$ for lint cleaners with covered condenser drums should be less than that for high-efficiency cyclones. Often in the literature, covered condenser drums and cyclones are assumed to have overall collection efficiencies of 50% and 90%, respectively. However, when these efficiencies are analyzed using lognormal PSDs for the dust and lognormal collection efficiency curves for the abatement devices; the percent $PM_{10}$ obtained by a cyclone is higher than the percent $PM_{10}$ obtained from a covered condenser drum. In other words, the mass of $PM_{10}$ and TSP will be higher for covered condenser drums than cyclones, but the percent $PM_{10}$ will be lower for the covered condenser drum than the cyclones. Therefore in this analysis, the 41.4% $PM_{10}$ value will be used for the lint cleaners with covered condenser drums as well as cyclones. In addition, the same anomaly occurs with the battery condenser, so the 35.9% $PM_{10}$ to TSP ratio were used for both the covered condenser drums and cyclones.

The MMD and true percent $PM_{10}$ were determined for all exhausts listed in the 1996 AP-42 using a trail and error procedure based on EPA's TSP and $PM_{10}$ cotton gin emission factors. In this procedure, PSD GSD values were assumed to be 1.8, 2.0, and 2.2. The results of the simulation are shown in Table 27. Mass median diameters ranged from 11.9 to 19.8 μm for all exhausts except for the trash process stream exhaust. The MMD associated with the trash exhaust ranged from 26.8 to 31.8 μm. True percent $PM_{10}$ values ranged from 4.7 to 40.1% for all process stream exhausts. Overall, the MMD and true percent $PM_{10}$ did vary between process stream exhausts. The effects of varying GSD from 1.8 to 2.2 were minimal in comparison to the exhaust type effects. Therefore, MMD and percent true $PM_{10}$ values based on an assumed GSD of 2.0 for all process would be sufficient in describing the PM emissions based on the 1996 AP-42 emission factors.

**Table 27.** Calculated MMD and true percent $PM_{10}$ values for selected cotton gin exhaust, based on the 1996 AP-42 list of cotton gin emission factors.

| Process Stream | GSD = 1.8 | | GSD = 2.0 | | GSD = 2.2 | |
|---|---|---|---|---|---|---|
| | MMD (μm) | $PM_{10}$ (%) | MMD (μm) | $PM_{10}$ (%) | MMD (μm) | $PM_{10}$ (%) |
| Unloading | 13.1 | 32.3 | 13.4 | 33.6 | 13.6 | 34.8 |
| 1st Stage Seed Cotton Cleaning | 15.7 | 22.1 | 16.2 | 24.3 | 16.7 | 25.8 |
| 2nd Stage Seed Cotton Cleaning | 13.9 | 28.8 | 14.2 | 30.7 | 14.5 | 31.9 |
| 3rd Stage Seed Cotton Cleaning | 15.2 | 23.8 | 15.6 | 26.1 | 16.1 | 27.3 |
| Distributor | 14.6 | 26.0 | 14.9 | 28.3 | 15.3 | 29.5 |
| Trash | 26.8 | 4.7 | 29.3 | 6.1 | 31.8 | 7.1 |
| Mote | 11.9 | 38.4 | 11.9 | 40.1 | 12.0 | 40.9 |
| Mote Trash | 18.0 | 15.9 | 18.9 | 17.9 | 19.8 | 19.3 |
| Lint Cleaning | | | | | | |
| (Covered Condenser Drum) | 13.1 | 32.3 | 13.4 | 33.6 | 13.6 | 34.8 |
| (Cyclones) | 13.1 | 32.3 | 13.4 | 33.6 | 13.6 | 34.8 |
| Battery Condenser | | | | | | |
| (Covered Condenser Drum) | 14.8 | 25.2 | 15.2 | 27.3 | 15.6 | 28.6 |
| (Cyclones) | 14.8 | 25.2 | 15.2 | 27.3 | 15.6 | 28.6 |

The mass density functions associated with the MMD and GSD values reported in Table 27 were weighted by the process stream exhaust 1996 AP-42 TSP emission factors divided by the total 1996 AP-42 TSP emission factor. The 3rd stage of seed cotton cleaning, mote trash, lint cleaners with cyclones, and battery condenser with cyclones were not included in the weighted average, as these are generally considered optional systems. The weighted mass density functions for each process stream exhaust were summed and a trail and error procedure was used to determine the weighted average values for MMD, GSD, and true percent $PM_{10}$ for assumed process stream exhaust GSD values of 1.8, 2.0, and 2.2. The results of the simulation are shown in Table 28. The average MMD values ranged from 15.0 to 16.0 μm. The average GSD values ranged from 1.90 to 2.34 and the true percent $PM_{10}$ ranged from 26.4 to 29.0%. The effects attributed to the assumed GSD values were minimal, based on the MMD and percent true $PM_{10}$ ranges. Therefore, a weighted average PSD based on an assumed GSD of 2.0 for all process stream exhausts would be sufficient in describing an average weighted PSD for cotton gins.

**Table 28.** Weighted average values for MMD, GSD, and true percent $PM_{10}$.

| Assumed process stream GSD | MMD (μm) | GSD | True $PM_{10}$ (%) |
|---|---|---|---|
| 1.8 | 15.0 | 1.90 | 26.4 |
| 2.0 | 15.5 | 2.12 | 28.0 |
| 2.2 | 16.0 | 2.34 | 29.0 |

The percent true $PM_{10}$ values reported in Table 28 were multiplied by the 1996 AP-42 TSP emission factors, in order to determine the corresponding true $PM_{10}$ emission factors based on an average weighted PSD. These values are shown in Table 29 along with the EPA 1996 AP-42 $PM_{10}$ emission factors. This table also includes $PM_{10}$ emission factors based on multiplying the true percent $PM_{10}$ values reported in Table 27 by the corresponding EPA 1996 AP-42 TSP emission factors.

The $PM_{10}$ emission factors varied by process stream exhaust, as expected. The $PM_{10}$ emission factor variation attributed to assumed process stream exhaust PSD GSD values of 1.8, 2.0, and 2.2 were minimal for the $PM_{10}$ emission factors based on individual process stream exhaust PSDs. These variations were also minimal for the $PM_{10}$ emission factors based on weighted average PSDs. The $PM_{10}$ emission factors based on true values were lower than the EPA 1996 AP-42 $PM_{10}$ emission factors. For example, the total $PM_{10}$ emission factors for the 8 selected process stream exhausts were 0.543, 0.831, and 0.387 kg/bale (1.2, 0.839, and 0.854 lb/bale) based on EPA's 1996 AP-42, individual process stream exhaust PSDs with an assumed GSD of 2.0, and average weighted PSDs with an assumed process stream exhaust PSD GSD of 2.0, respectively. The lint cleaner, battery condenser, and trash process stream exhaust $PM_{10}$ emission factors were affected more than the other process stream exhaust in adjusting the EPA 1996 AP-42 emission factors to reflect true $PM_{10}$. Although, the total true $PM_{10}$ emission factors based on individual process stream exhaust PSDs and average weighted PSDs are similar, the $PM_{10}$ emission factors for the lint cleaners and trash process stream exhausts did vary. For example, the $PM_{10}$ emission factor for the trash stream based on individual process stream PSDs was 0.015 kg/bale (0.033 lb/bale) and the corresponding emission factor for the weighted average PSD was 0.069 kg/bale (0.151 lb/bale).

**Table 29.** AP-42 $PM_{10}$ emission factors in kg/bale (lb/bale) and calculate factors based on individual process steam exhaust PSDs and weighted average PSDs for various assumed GSDs.

| Process Stream | 1996 AP-42 | Individual process stream PSD | | | Average PSD | | |
|---|---|---|---|---|---|---|---|
| | | GSD 1.8 | GSD 2.0 | GSD 2.2 | GSD 1.8 | GSD 2.0 | GSD 2.2 |
| Unloading | 0.054 | 0.043 | 0.044 | 0.046 | 0.035 | 0.037 | 0.038 |
| | (0.12) | (0.094) | (0.097) | (0.101) | (0.077) | (0.081) | (0.084) |
| 1st Stage Seed Cotton Cleaning | 0.054 | 0.036 | 0.040 | 0.042 | 0.043 | 0.046 | 0.047 |
| | (0.12) | (0.080) | (0.087) | (0.093) | (0.095) | (0.101) | (0.104) |
| 2nd Stage Seed Cotton Cleaning | 0.042 | 0.031 | 0.034 | 0.035 | 0.029 | 0.030 | 0.032 |
| | (0.093) | (0.069) | (0.074) | (0.077) | (0.063) | (0.067) | (0.070) |
| 3rd Stage Seed Cotton Cleaning | 0.015 | 0.010 | 0.011 | 0.012 | 0.011 | 0.012 | 0.013 |
| | (0.033) | (0.023) | (0.025) | (0.026) | (0.025) | (0.027) | (0.028) |
| Distributor | 0.012 | 0.008 | 0.009 | 0.010 | 0.009 | 0.009 | 0.010 |
| | (0.026) | (0.018) | (0.020) | (0.021) | (0.019) | (0.020) | (0.021) |
| Trash | 0.034 | 0.011 | 0.015 | 0.017 | 0.065 | 0.069 | 0.071 |
| | (0.074) | (0.025) | (0.033) | (0.038) | (0.143) | (0.151) | (0.157) |
| Mote | 0.059 | 0.049 | 0.051 | 0.052 | 0.034 | 0.035 | 0.037 |
| | (0.13) | (0.108) | (0.112) | (0.115) | (0.074) | (0.078) | (0.081) |
| Mote Trash | 0.095 | 0.005 | 0.006 | 0.007 | 0.009 | 0.010 | 0.010 |
| | (0.21) | (0.012) | (0.014) | (0.015) | (0.020) | (0.022) | (0.022) |
| Lint Cleaning | | | | | | | |
| (Covered Condenser Drum) | 0.250 | 0.161 | 0.169 | 0.174 | 0.132 | 0.140 | 0.145 |
| | (0.55) | (0.355) | (0.370) | (0.383) | (0.290) | (0.308) | (0.319) |
| (Cyclone) | 0.109 | 0.085 | 0.089 | 0.092 | 0.069 | 0.074 | 0.076 |
| | (0.24) | (0.187) | (0.195) | (0.202) | (0.153) | (0.162) | (0.168) |
| Battery Condenser | | | | | | | |
| (Covered Condenser Drum) | 0.039 | 0.020 | 0.021 | 0.022 | 0.020 | 0.022 | 0.022 |
| | (0.085) | (0.043) | (0.046) | (0.049) | (0.045) | (0.048) | (0.049) |
| (Cyclone) | 0.006 | 0.005 | 0.005 | 0.005 | 0.005 | 0.005 | 0.005 |
| | (0.014) | (0.010) | (0.011) | (0.011) | (0.010) | (0.011) | (0.011) |
| Total[a] | 0.543 | 0.359 | 0.381 | 0.398 | 0.366 | 0.387 | 0.402 |
| | (1.198) | (0.792) | (0.839) | (0.877) | (0.806) | (0.854) | (0.885) |

[a] Total does not include values from the 3rd stage of seed cotton cleaning, mote trash, lint cleaners with cyclones, or the battery condenser with cyclones.

# SUMMARY AND CONCLUSIONS

There are several errors associated with the current air pollution rules and regulations established by the EPA which should be minimized to assure equal regulation of air pollutants between and within all industries. Potentially, one of the most significant errors is attributed to the interaction of the industry specific PSD and sampler performance characteristics. Currently, the regulation of PM is based on sampler measurements and NOT true concentrations.

A sampler concentration corresponds to the concentration collected by a PM sampler. This concentration is dependent on the sampler's performance characteristics (i.e. $d_{50}$ and slope). Since the concentration is based on the sampler's performance characteristics; there are two inherent errors associated with the measurement. For a $PM_{10}$ sampler, the first error corresponds to the mass of particles less than 10 μm that should have been capture on the filter but was removed from the air stream by the pre-separator. The second error (for a $PM_{10}$ sampler) corresponds to the mass of particles greater than 10 μm that should have been removed from the air stream by the pre-separator but was allowed to pass through the pre-separator and be captured on the filter. When the MMD of the dust being sampled is less than the sampler $d_{50}$; under-sampling of the mass of particles less than 10 μm occurs. When the MMD of the dust being sampled is equal to the $d_{50}$ of the sampler; the sampler provides a measurement equivalent to the true mass of particles less than 10 μm. When the MMD of the dust being sampled is greater than the $d_{50}$ of the sampler; over-sampling of the mass of particles less than 10 μm occurs. These types of errors are also associated with the ambient $PM_{2.5}$, $PM_{10}$ stack, and the calculation of $PM_{coarse}$.

A true concentration refers to the mass of particles less than the size of interest. In order for a sampler to provide a true concentration, independent of the MMD of the dust being sampled, the sampler would have to maintain a slope of 1.0 (i.e. the sampler's penetration curve would be represented by a step function).

According to the literature, EPA decided to regulate $PM_{2.5}$ based on the availability of the dichotomous sample. The $PM_{2.5}$ regulation was not based on determining the resiprable fraction of PM. ACGIH, ISO, and others have defined the resiprable fraction of PM as having a cutpoint between 3.5 and 5 $\mu$m. The final justification for using true $PM_{2.5}$ values as opposed to sampler based concentration comes from the literature in the following direct quote:

> Staff also recommended the use of a sharp 2.5 micron cutpoint for a fine particle indicator. $PM_{2.5}$ does have some potential for intrusion of the tail of the coarse mode during episodes of fugitive dust concentrations. Staff recommends a sharp inlet for the FRM to minimize this potential intrusion of coarse mode particles. Such intrusion into $PM_{2.5}$ measurement is not anticipated to be significant in most situations; nevertheless, if subsequent data reveal problems in this regard, this issue can and should be addressed on a case-by-case basis in the monitoring and implementation programs. Because the purpose of a $PM_{2.5}$ standard is to direct controls toward sources of fine mode particles, it would be appropriate to develop analytical procedures for identifying those cases where a $PM_{2.5}$ standard violation would not have occurred in the absence of coarse mode particle intrusion. Consideration should be given to a policy similar to the natural events policy for addressing such cases. (USEPA, 1996c)

When EPA implemented the $PM_{10}$ NAAQS and approved the FRM ambient air sampler; EPA's intent was for the performance characteristics of the $PM_{10}$ sampler to mimic the thoracic penetration curve. This standard corresponds to PM in the ambient air (i.e. not impacted by only one source). Therefore, the question becomes "is it appropriate to use EPA approved ambient $PM_{10}$ samplers or $PM_{10}$ stack samplers, which have performance that are more loosely defined than ambient air samplers, to determine emission values for individual source". If the issue pertains to determining emission rates from cotton gin exhausts or from farming operations, then the answer is absolutely NOT. In general, these types of operations are emitting fairly large PM relatively close to the ground, which means that a large majority of the emitted PM will settle out of the air within a short time frame within a short distance. Now as with any general statement, there will be exceptions; such as high wind events. Therefore, many of the ambient PM monitoring stations will never be exposed to large particles emitted by these operations.

So how is this a problem? If a state or air district finds itself in non-attainment with the $PM_{10}$ NAAQS, then the corresponding agencies will, most likely, be required to reduce $PM_{10}$ emissions within the air shed. In order to reduce emissions from individual sources, the amount of PM emitted by the sources must be known or estimated. This is typically accomplished through source sampling or the use of emission factors, which may have been determined from source sampling or interrupted by some other means. In order to illustrate why it is crucial that emission factors, emission rates, and/or emission concentration from individual sources be based on true $PM_{10}$ and not $PM_{10}$ sampler measurements, the following example is provided.

Assume that Method 201a samplers were setup to monitor two commercial operations. Assume that the samplers have performance characteristics described by a $d_{50}$ of 11 μm and a slope of 1.76 (both parameters are within the performance criteria defined by EPA). Now assume that one operation is a power plant and is emitting PM (sampled by the $PM_{10}$ sampler) that can be described by a lognormal distribution with a MMD of 5 μm and a GSD of 1.5. Assume that operation the second operation is an agricultural operation and is emitting PM (sampled by the $PM_{10}$ sampler) that can be described by a lognormal distribution with a MMD of 20 μm and a GSD of 1.5. Further, assume that the $PM_{10}$ sampler used to monitor each of the operations measures 100 $\mu g/m^3$. Now based on the methods laid out in this research, the true PM (PM less than 10 μm) emitted from each industry would be defined as follows.

Based on a PSD analysis, the percent of PM mass less than 10 μm being emitted from the power plant is 96%. Based on the PSD and sampler performance characteristics the percent of TSP capture by the $PM_{10}$ sampler is 87%. The TSP concentration emitted from the power plant is 115 $\mu g/m^3$ (i.e. 100 $\mu g/m^3$ (PM measured by the $PM_{10}$ sampler)/0.87 (% of TSP captured by the $PM_{10}$ sampler). Multiplying the TSP concentration by the true fraction of $PM_{10}$ (i.e. 115 $\mu g/m^3$ * 0.96), the true $PM_{10}$

concentration is determined to be 110 μg/m$^3$. Therefore, the PM$_{10}$ sampler under-estimated the true PM$_{10}$ concentration by 9%.

For the agricultural operation, using the previous procedures, the true percent PM$_{10}$ is 4.37% and based on the PSD and sampler performance characteristics the percent of TSP captured by the PM$_{10}$ sampler is 19.5%. Therefore, the TSP concentration is 513 μg/m$^3$. Resulting in the true PM$_{10}$ emitted from operation being equal to 22.4 μg/m$^3$. In this case, the PM$_{10}$ sampler over-estimated the true PM$_{10}$ by 346%.

Based on this scenario, the two operations are not being equally regulated (i.e. 110% of the PM emitted from the power plant and measured by the PM$_{10}$ sampler corresponds to PM less than 10 μm; whereas only 22% of the PM from the agricultural operation and measured by the PM$_{10}$ sampler corresponds to PM less than 10 μm).

Looking at the issue from another viewpoint, assume that the two operations described previously are emitting 100 μg/m$^3$ true PM$_{10}$. If the percent of TSP measured by the PM$_{10}$ sampler (defined as measured PM$_{10}$) is divided by the percent of true PM$_{10}$ and this ratio is multiplied by the true concentration of PM$_{10}$ being emitted, the PM$_{10}$ sampler concentrations can be determined. For the power plant, the PM$_{10}$ sampler could measure a concentration as high as 91 μg/m$^3$. For the agricultural operation, the PM$_{10}$ sampler could measure a concentration as high as 446 μg/m$^3$. What this means for the agricultural operation is that 100 μg/m$^3$ of the PM being sampled is less than 10 μm and 346 μg/m$^3$ of the PM being sampled is larger than 10 μm. Further, depending on the release height and meteorology conditions, the majority of this 346 μg/m$^3$ of particles larger than 10 μm will rapidly settle out of the air. Therefore, the answer to the question of "is it appropriate to use PM$_{10}$ samplers to determine PM$_{10}$ emission values for agricultural operation" is absolutely not.

The following are generalized conclusions drawn from this research:

- if MMD < $d_{50}$ then $C_{sampler} < C_{true}$;
- if MMD = $d_{50}$ then $C_{sampler} = C_{true}$;

- if MMD > $d_{50}$ then $C_{sampler} > C_{true}$;
- as GSD increases the concentration ratio of $C_{sampler}$ to $C_{true}$ decreases;
- as sampler slope decreases the concentration ratio of $C_{sampler}$ to $C_{true}$ decreases;
- the MMD of PM emitted by cotton gin exhaust will vary by location and processing stream;
- the MMD of cotton gin exhausts are larger than 10 μm; and
- current $PM_{10}$ emission factors associated with cotton gin exhausts are generally over-estimated because of inherent sampler errors.

Results of the analysis presented in this research show that not all industries are being equally regulated in terms of PM and that **ALL** industries should be concerned with the current site-specific regulations implemented by EPA and enforced by SAPRA’s.

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# APPENDIX A

# JUSTIFICATION FOR USING DRY STANDARD (VS. ACTUAL) FLOW VOLUMES WHEN COMPARING PARTICULATE MATTER SAMPLER MEASUREMENTS TO PARTICULATE MATTER STANDARDS

The literature suggests that health effect related issues are the primary reasons for reporting mass concentrations in terms of actual unit volume. According to the USEPA (1996) "the rationale for aerosol sampling is to mimic respiratory penetration (occurring at local conditions), a correction after the fact may not be appropriate". Health effect studies have been conducted in cool and warm climates, and in cities at high altitudes (e.g. Denver) as well as near sea level (e.g. Philadelphia). Results from these studies have shown no evidence that the risks associated with PM exposures are affected by variations in altitude. These reports further suggest that adjusting temperature and pressure to dry standard conditions would not significantly change the reported concentration and would be below the detection limits of epidemiological studies. Although the delivery dose of PM might be expected to increase at extreme altitudes, for those not acclimatized to such locations, dosimetry studies have provided no clear support justifying any PM concentration adjustment to standard conditions.

The following example illustrates why PM concentration should be reported in terms of mass per dry standard cubic meter, μg/dscm. The amount of oxygen required by living organisms is based on the mass of oxygen and not the volume of oxygen. For this example, 0.45 kg (1 lb) of oxygen per minute is assumed to be the mass of oxygen required by humans at a normal activity level. Approximately 21% of the air is composed of oxygen. Therefore, humans would require 2.18 kg air/min (4.8 lb air/min), i.e. (0.45 lb $O_2$/min) * (1 kg air/0.21 kg $O_2$). In this example, a comparison is made between a person located in Houston, TX (an elevation near sea level) and a person located in Lubbock, TX (an elevation of 974 m (3,195 feet) above sea level). Assuming that the relative humidity at both locations is 0% and the ambient temperature is 21$^{o}$C (70$^{o}$F), then the air density at Houston and Lubbock would be 1.2 kg air/actual cubic meter (0.075 lb air/actual cubic foot) and 1.0 kg air/actual cubic meter (0.0625 lb air/actual cubic foot), respectively. By multiplying the mass of air required by the inverse of air density, air volume intake can be determined. Therefore, the person in Houston, TX will require 1.81 actual cubic meters of air per minute (64 actual cubic feet of air per minute), in order to meet the oxygen requirements. Likewise, the person in

Lubbock, TX will require 2.18 actual cubic meter of air per minute (77 actual cubic feet of air per minute).

According to the EPA guidelines for high volume sampling, the target high-volume sampler flow rate is 1.13 actual cubic meters per minute (40 actual cubic feet per minute). The ambient $PM_{10}$ concentrations in both Houston and Lubbock, TX were assumed to be 150 $\mu g/m^3$. Based on these assumptions, how much $PM_{10}$ are the people in both locations actually intaking? This can be calculated by multiplying the air volume requirement by the $PM_{10}$ concentration. Based on these calculations, the person in Lubbock, TX would intake 326 μg/min, as compared to 271 μg/min for the person in Houston, TX. Therefore, people in Lubbock, TX are subject to higher rates of PM exposure than people in Houston, TX even though the $PM_{10}$ sampler measurements showed that exposure levels were identical for both locations.

So how is this issue corrected? Assume that the PM regulations are based on mass per dry standard cubic meter. Assuming that $PM_{10}$ concentrations for both locations were 150 μg/dscm, then the actual $PM_{10}$ concentrations for both locations could be calculated by multiplying the reported concentration by the inverse of the air density. Using this procedure, the actual $PM_{10}$ concentrations for Houston and Lubbock, TX would be 150 and 125 μg/acm. Performing the calculations previously described for determining potential intake, the person in Houston, TX would intake 271 μg/min and the person in Lubbock, TX would intake 271 μg/min. Therefore, people at both locations are exposed to the same levels of PM. Based on this example and regardless of the epidemiology study results, the PM regulations should be regulated in terms of mass per dry standard cubic meter.

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# APPENDIX B

## THEORETICAL BASIS FOR USING TSP LOADED FILTERS FOR PARTICLE SIZE ANALYSIS

The following scenario describes some of the theoretical aspects associated with conducting particle size analysis of particulate matter (PM) captured on a total suspended particulate (TSP) sampler filter. In this scenario, several broad assumptions are made. First, the PM in the ambient air is assumed to be characterized by a lognormal particle size distribution (PSD), described by a mass median diameter (MMD) of 25 μm and a geometric standard deviation (GSD) of 2.0. Secondly, the TSP sampler performance characteristics are assumed to be described by a cumulative lognormal penetration curve, described by a $d_{50}$ of 45 μm and a slope of 2.0. The graph shown in Figure 69 illustrates how a PSD of dust in the ambient air and PSD of PM captured on the TSP sampler filter theoretically compare. The following are some key points associated with this comparison:

1) 2.28% of the PM mass associated with particles in the ambient air are greater than 100 μm;
2) 9.31% of the PM mass associated with particles in the ambient air are less than 10 μm;
3) 0.17% of the PM mass associated with particles captured by the TSP sampler are greater than 100 μm;
4) 12.75% of the PM mass associated with particles captured by the TSP sampler are less than 10 μm; and
5) the TSP sampler only captures 72.6% of the PM mass associated with particles in the ambient air.

The following example explains the key points in greater detail. The concentration of PM in the ambient air is assumed to be 100 $\mu g/m^3$ and the PSD of the PM in the ambient air is assumed to be characterized by a MMD of 25 μm and a GSD of 2.0. The PM in the ambient air is assumed to be sampled by a TSP sampler with performance characteristics described by a $d_{50}$ equal to 45 μm and a slope of 2.0. Based on the key points, the TSP sampler would measure 72.6 $\mu g/m^3$ (i.e. the TSP sampler captures 72.6% of the PM in the ambient air and this value is multiplied by the concentration, 100 $\mu g/m^3$, of PM in the ambient air). The concentration of true $PM_{10}$ in the ambient air is 9.31 μg/m$^3$ (i.e. 9.31% of the PM mass associated with particles in the ambient air less than 10 μm). The particle size analysis of the TSP filter would theoretically show that 12.75% of the PM sampled is less than 10 μm. By multiplying this percentage by the concentration determined from the TSP sampler (i.e. 0.1275 * 72.6), the concentration of particles less than 10 μm would be 9.26 μg/m$^3$. Therefore in this example, the TSP sampler and particle sizing procedure would result in a 0.05 μg/m$^3$ under-estimation (0.54%) of the true PM$_{10}$ concentration. On the other hand, if a PM$_{10}$ sampler with a d$_{50}$ of 10 μm and a slope of 1.5 were used to determine the PM$_{10}$ concentration, the resulting concentration would be 12.7 μg/m$^3$. The PM$_{10}$ sampler concentration would over-estimate the true PM$_{10}$ concentration by about 37%. Further, if the PM$_{10}$ sampler were characterized by a d$_{50}$ of 10.5 μm and a slope of 1.6 the resulting PM$_{10}$ concentration would be 15 μg/m$^3$. This would correspond to a 61% over-estimation of the true PM$_{10}$ concentration.

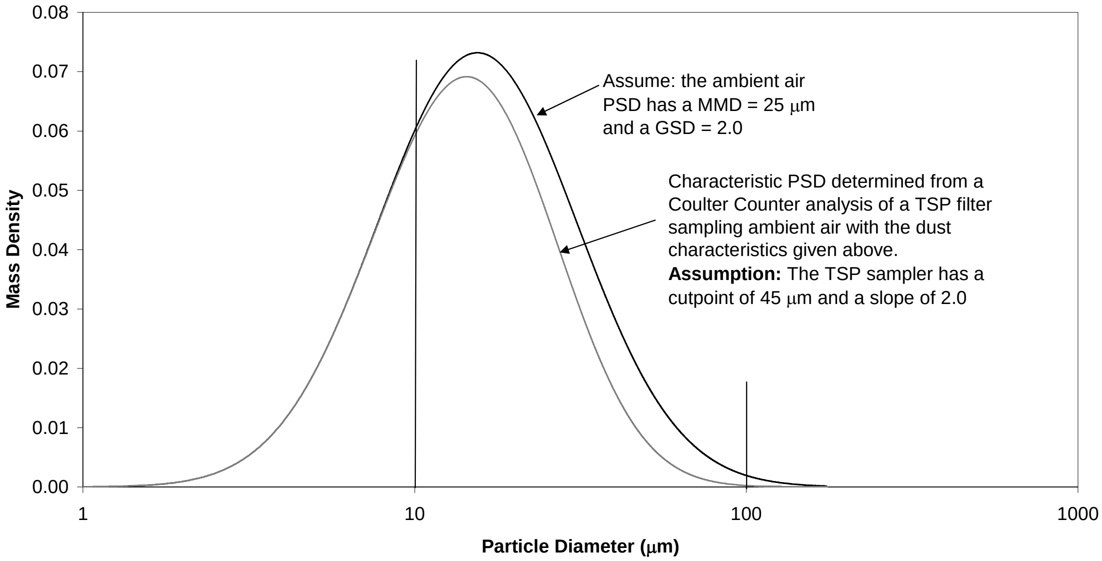

**Figure 69.** Theoretical comparison of an ambient air PSD of PM, described by a MMD of 25 μm and a GSD of 2.0, to the PSD of PM capture on a TSP sampler, described by a d$_{50}$ of 45 μm and a slope of 2.0, filter when sampling the ambient air.

# APPENDIX C

# EVALUATION OF FILTER MEDIA AND METHODS USED IN DISPERSING FILTER CAPTURED PM IN ELECTROLYTE FOR COULTER COUNTER ANALYSIS

Filter media, used in PM sampling, is composed of particles and fibers that can adversely impact particle size analyses. Raina and Parnell (1995) evaluated blank (unused) glass fiber and poly-web filters to determine the background counts associated with the filter media. According to Raina and Parnell (1995), the particle size analysis of the glass fiber filter media produce a particle count of 200,000 particles with a MMD of 5.2 μm and a standard deviation of 0.7 when 2 cc of solution was analyzed. A similar analysis conducted on poly-web filter media resulted in a particle count of 44,941 with a MMD of 18 μm and a standard deviation of 0.9. Herber (1988) conducted a similar study. Herber (1988) reported that the glass fiber filter media particle count was 162,233 when 6 cc of solution was analyzed and the poly-web filter media particle count was 81,904 when 40 cc of solution was analyzed. Results from theses studies are listed in Table 30.

**Table 30.** Previous comparison of particle background analyses of filter media.

| | Raina and Parnell (1995) | | Herber (1988) | | |
|---|---|---|---|---|---|
| Filter Type | Glass Fiber | Poly-Web | Glass Fiber | Quartz | Poly-Web |
| Sample size in 20 ml dispersant | 1”x2” | 1”x4” | 1”x2” | 1”x2” | 1”x6” |
| Amount added to 10 ml electrolyte | 2 cc | 2 cc | 6cc | 5cc | 40cc |
| Median (μm) | 5.2 | 18.0 | NR | NR | NR |
| Standard Deviation | 0.7 | 0.9 | NR | NR | NR |
| Particles Counted | 200,000 | 44,941 | 162,233 | 174,384 | 81,904 |
| Sampling Time (min) | 2.8 | 10.9 | 2.6 | 3.5 | 7.6 |
| % of particulate <= 10 micron | 75.2 | 33.4 | 71 | 64 | 21 |

NR – Not Reported

A study was conducted to evaluate the background particle characteristics of various filter media using the Coulter Counter Multisizer III. Filter media evaluated included a glass fiber filters used with the TSP high-volume sampler, glass fiber filters used with the EPA Method 5 stack sampler, glass fiber filters used with the EPA Method 201a stack sampler, poly-web filters, 2 μm Teflon filters, 0.45 μm Teflon filters, 0.2 μm

Teflon filters, polyvinyl-chloride filters, cotton swabs, foam swabs, nylon swabs, and nylon swabs rolled across a high-volume TSP sampler glass fiber filters. In this study, the filter media was cut or directly submersed in 40 ml of electrolyte. This solution was subjected to a ultrasonic bath for 15 minutes. The solution was gradually added to the electrolyte contained in the Coulter Counter beaker, until the concentration in the beaker was approximately six percent. The Coulter Counter was configured to analyze 500 µl of solution. Three replications with three repeated measures were completed for all treatments. The PROC Mixed (Littell, et al., 1996) procedure in SAS was utilized to compare the filter media background characteristics. Results of the test are listed in Table 31. The background count associated with the high-volume TSP sampler glass fiber filter media was significantly higher than all other filter media tested. This filter media had a particle count of 12,388 with a MMD of 5.6 µm. The background count associated with the EPA Method 5 glass fiber filter media was significantly different than all other filter media. The filter media had a particle count of 8,382 with a MMD of 5.1 µm. The background count associated with the EPA Method 201a glass fiber filter was also significantly different than all other filter media evaluated. The background count was 4,980 with a MMD of 5.3 µm. The background count for the polyvinyl-chloride filter was significantly different than all other filter media evaluated. The background count was 969 with a MMD of 15.8 µm. There were no other significant particle count differences detected between other treatments. Particle counts for all other treatments were lower than 248. Table 33 lists the treatments and figure numbers associated with the number and volume distributions for each of the treatments.

A secondary test was conducted to determine the best method of transferring PM captured on a high-volume TSP sampler glass fiber filter to the electrolyte (i.e., because of the large particle background count associated with the glass fiber filter media a method was need to transfer the PM collected on the glass fiber filter to the electrolyte). The large glass fiber filters were loaded with cornstarch using an air-wash. The methods tested included: cutting a section from the large glass fiber filter, subtracting a background PSD obtained from blank filters from the PSD of the loaded filter, rolling a nylon swab across a loaded glass fiber filter, and transferring the PM captured on a glass fiber filter to a poly-web filter by inverting the glass fiber filter and pulling air across the filters (the same procedure was performed for 2 μm and 0.2 μm Teflon filters). Three replications and three repeated measures were completed for each treatment. Results from the test are listed in Table 32. Table 33 lists the figure numbers associated with the number and volume distributions associated with each treatment. The PROC Mixed (Littell, et al., 1996) procedure in SAS was utilized to compare the treatments. Several significant differences were detected between the PSD characteristics of the various treatments. Based the MMD, $d_{15.9}$, $d_{84.1}$, and percent less than 10 μm, rolling the nylon swab across the cornstarch loaded glass fiber filter appeared to produce the best overall results. Based on this analysis, all glass fiber filters received from the stack sampling tests were analyzed by rolling the nylon swab across the filter and submersing the swab in an electrolyte solution. If given the option, stack sampling should be conducted using Teflon filters.

**References:**

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Raina, M. and C.B. Parnell, Jr. 1994. Determination of PM-10 concentrations from TSP data using the Coulter Counter Multisizer. ASAE Paper No. 944537. St. Joseph, MI: ASAE.

**Table 31.** Particle size and count characteristics associated with various filter media as determined by the Coulter Counter Multisizer III, based on 500 μL samples.

| Media | MMD (μm) | $D_{15.9}$ (μm) | $D_{84.1}$ (μm) | % less than 10 μm | Particle Count |
|---|---|---|---|---|---|
| TSP high-volume TSP sampler glass fiber filter | 5.6 b | 3.1 cd | 10.1 f | 82.1 a | 12,388 a |
| EPA Method 5 glass fiber filter | 5.1 b | 2.9 d | 11.0 ef | 83.1 a | 8,382 b |
| EPA Method 201a glass fiber filter | 5.3 b | 2.9 cd | 13.6 def | 79.8 a | 4,980 c |
| Polyweb filter | 21.5 a | 9.1 abcd | 28.5 a | 22.7 cde | 231 e |
| 2 μm Teflon filter | 18.7 a | 10.9 abc | 21.8 abc | 34.5 bcd | 114 e |
| 0.45 μm Teflon filter | 17.8 a | 10.4 abcd | 19.9 bcd | 20.0 de | 69 e |
| 0.2 μm Teflon filter | 16.6 a | 9.6 abcd | 19.4 bcde | 16.9 e | 58 e |
| Polyvinyl-chloride filter | 15.8 a | 5.4 bcd | 27.7 ab | 32.7 bcde | 969 d |
| Cotton swab | 22.4 a | 13.6 a | 24.6 abc | 27.3 bcde | 95 e |
| Foam swab | 19.0 a | 12.6 ab | 20.6 abcd | 27.7 bcde | 49 e |
| Nylon swab | 14.9 a | 7.2 abcd | 16.4 cdef | 43.2 b | 65 e |
| Nylon swab rolled across a TSP high-volume sampler glass fiber filter | 15.5 a | 6.2 abcd | 23.2 abc | 36.6 bc | 248 e |

Means in the same column followed by the same letter are not statistically different at the 0.05 level of significance.

**Table 32.** Particle size and count characteristics associated with various techniques tested to limit background contamination associated with dispersing PM captured on a TSP high-volume sampler glass fiber filter into electrolyte as determined by the Coulter Counter Multisizer III, based on 500 μL samples.

| Technique | MMD (μm) | $D_{15.9}$ (μm) | $D_{84.1}$ (μm) | % less than 10 μm |
|---|---|---|---|---|
| Cornstarch | 15.6 a | 11.3 b | 19.6 a | 10.0 d |
| Cut section from a cornstarch loaded TSP high-volume sampler glass fiber filter | 14.8 b | 10.2 e | 19.5 ab | 15.1 a |
| Background PSD subtracted from a cornstarch loaded TSP high-volume sampler glass fiber filter | 14.9 b | 10.7 d | 19.0 abc | 12.7 b |
| Nylon swab rolled across a cornstarch loaded TSP high-volume sampler glass fiber filter | 15.7 a | 11.9 a | 19.5 ab | 7.0 e |
| Transferring cornstarch from a loaded TSP high-volume sampler glass fiber filter to a polyweb filter | 15.0 b | 10.9 cd | 19.3 abc | 11.3 c |
| Transferring cornstarch from a loaded TSP high-volume sampler glass fiber filter to a 2μm Teflon filter | 15.0 b | 11.1 bc | 18.7 c | 10.7 cd |
| Transferring cornstarch from a loaded TSP high-volume sampler glass fiber filter to a 0.2 μm Teflon filter | 14.7 b | 10.7 d | 18.7 bc | 12.0 bc |

Means in the same column followed by the same letter are not statistically different at the 0.05 level of significance.

**Table 33.** Figures associated with the volume and number based average Coulter particle size distributions for the various filter media and methods tested.

| Media/Test Method | Volume Distribution | Number Distribution |
|---|---|---|
| ***Media*** | Figure 70 | Figure 71 |
| TSP high-volume sampler glass fiber filter | Figure 72 | Figure 73 |
| EPA Method 5 glass fiber filter | Figure 74 | Figure 75 |
| EPA Method 201a glass fiber filter | Figure 76 | Figure 77 |
| Polyweb filter | Figure 78 | Figure 79 |
| 2 μm Teflon filter | Figure 80 | Figure 81 |
| 0.45 μm Teflon filter | Figure 82 | Figure 83 |
| 0.2 μm Teflon filter | Figure 84 | Figure 85 |
| Polyvinyl-chloride filter | Figure 86 | Figure 87 |
| Cotton swab | Figure 88 | Figure 89 |
| Foam swab | Figure 90 | Figure 91 |
| Nylon swab | Figure 92 | Figure 93 |
| Nylon swab rolled across a TSP high-volume sampler glass fiber filter | Figure 94 | Figure 95 |
|  | Figure 96 | Figure 97 |
| ***Methods*** | Figure 98 | Figure 99 |
| Cornstarch | Figure 100 | Figure 101 |
| Cut section from a cornstarch loaded TSP high-volume sampler glass fiber filter | Figure 101 | Figure 103 |
| Background PSD subtracted from a cornstarch loaded TSP high-volume sampler glass fiber filter | Figure 104 | Figure 105 |
| Nylon swab rolled across a cornstarch loaded TSP high-volume sampler glass fiber filter | Figure 106 | Figure 107 |
| Transferring cornstarch from a loaded TSP high-volume sampler glass fiber filter to a polyweb filter | Figure 108 | Figure 109 |
| Transferring cornstarch from a loaded TSP high-volume sampler glass fiber filter to a 2μm Teflon filter | Figure 110 | Figure 111 |
| Transferring cornstarch from a loaded TSP high-volume sampler glass fiber filter to a 0.2 μm Teflon filter | Figure 112 | Figure 113 |

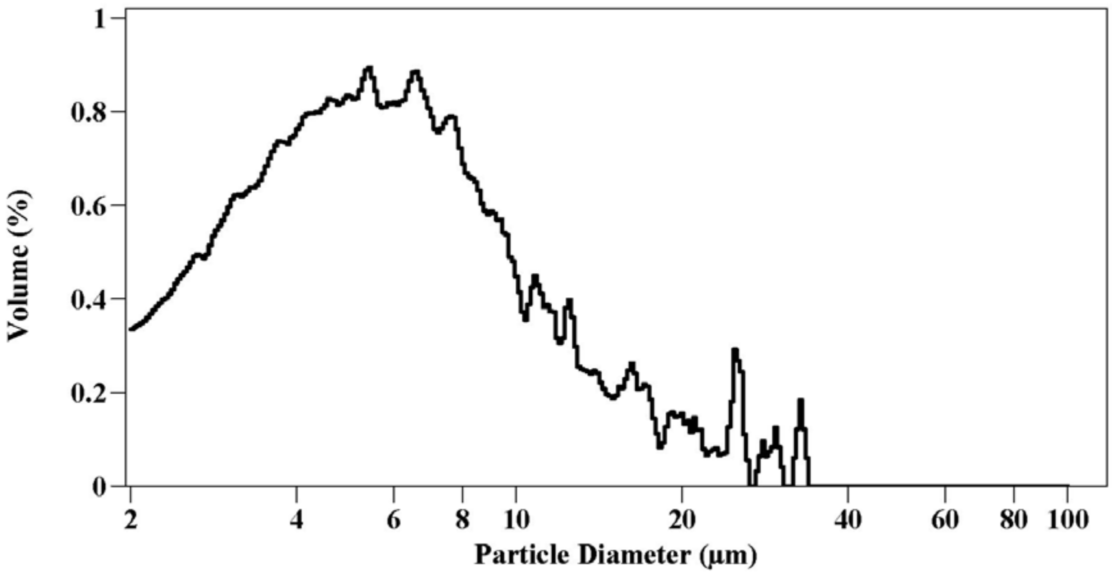

**Figure 70.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank TSP high-volume sampler glass fiber filter.

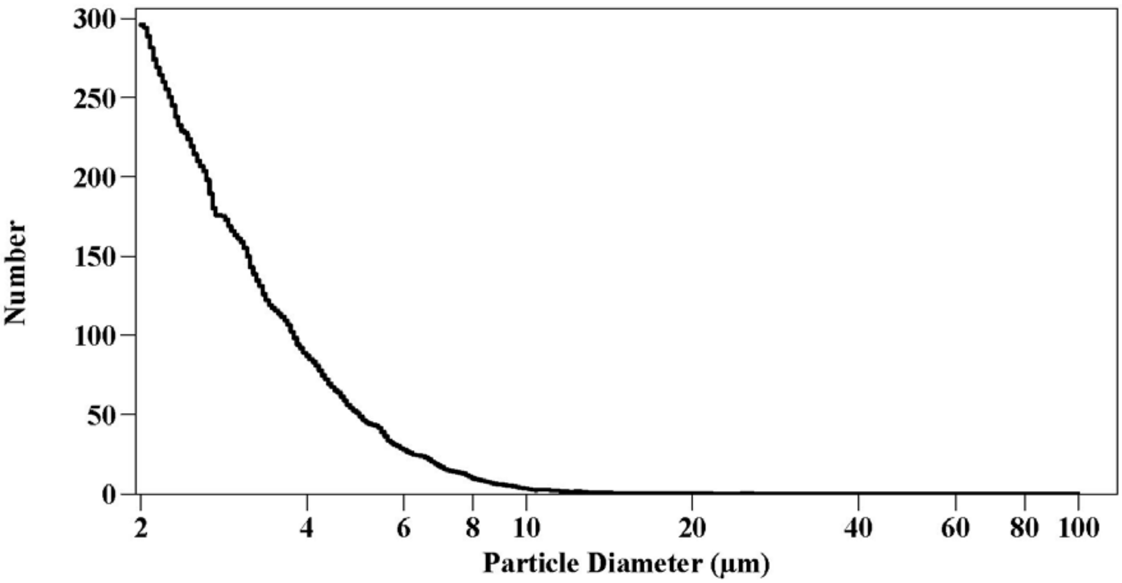

**Figure 71.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank TSP high-volume sampler glass fiber filter.

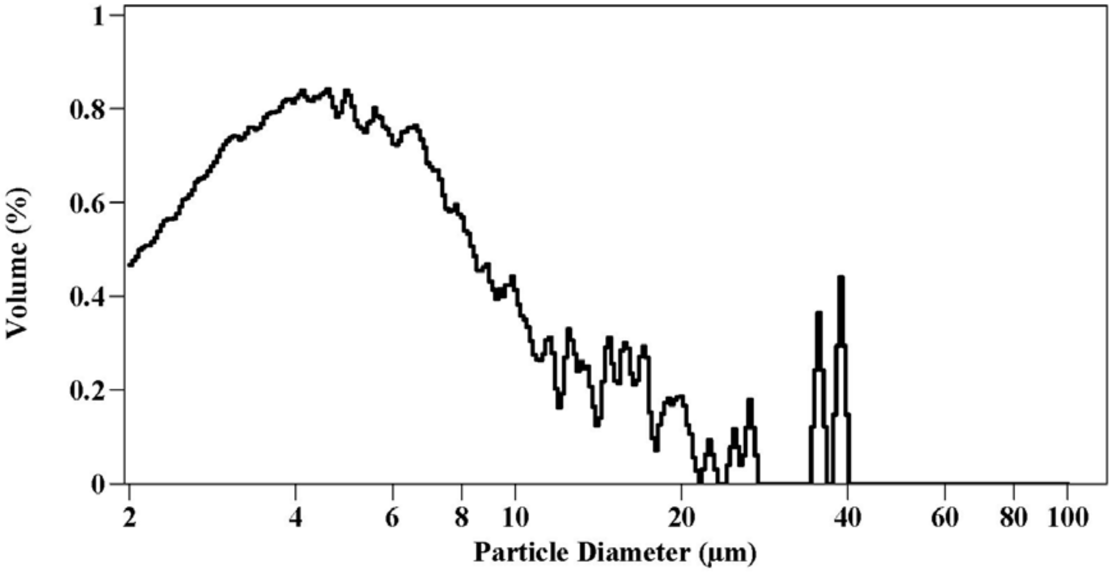

**Figure 72.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank EPA Method 5, TSP stack sampler glass fiber filter.

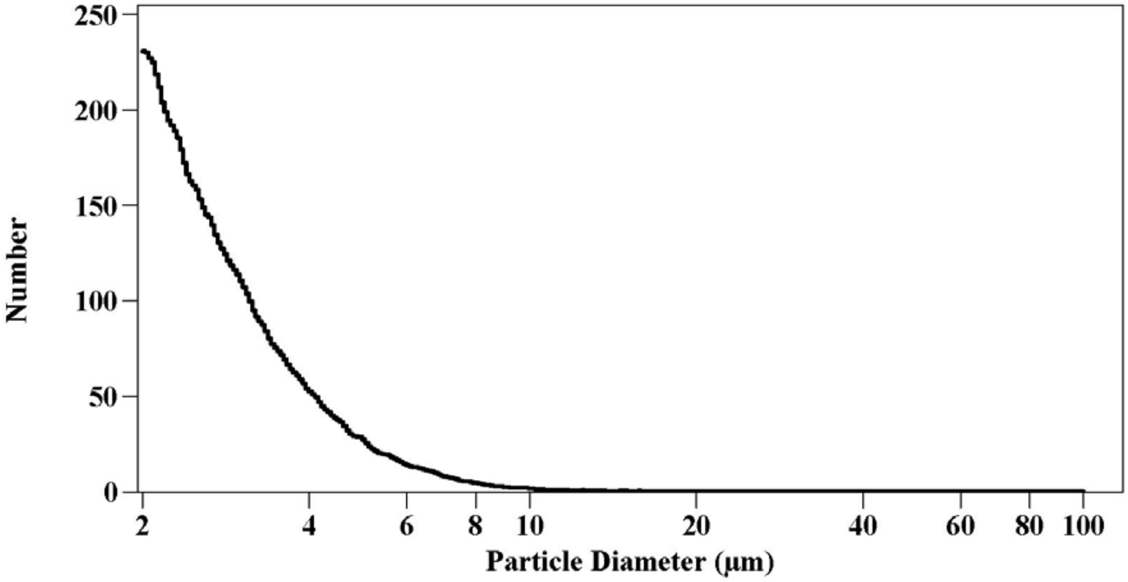

**Figure 73.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank EPA Method 5, TSP stack sampler glass fiber filter.

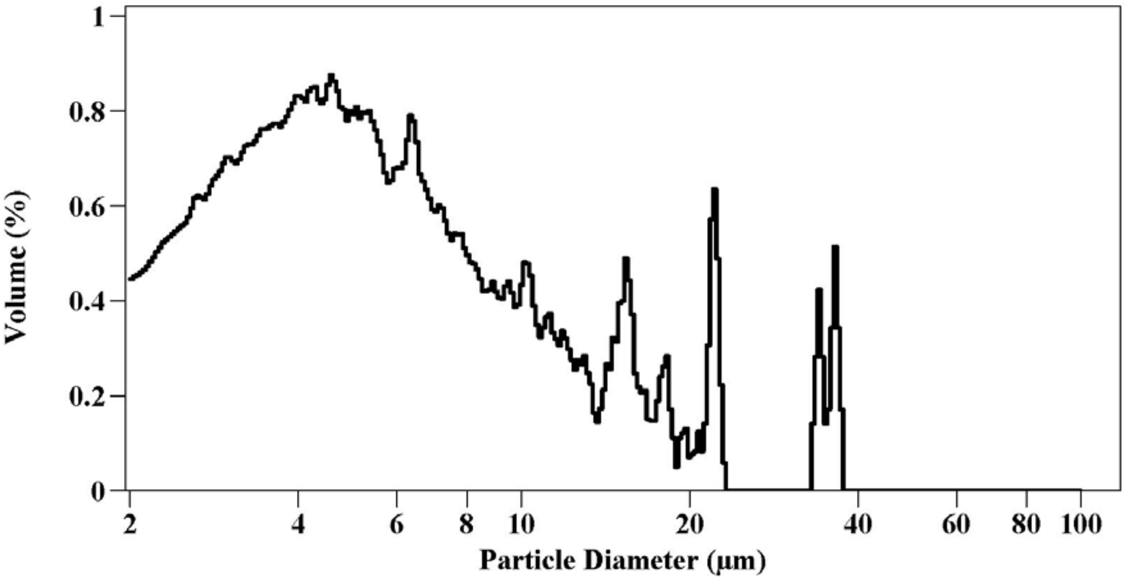

**Figure 74.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank EPA Method 201a, $PM_{10}$ stack sampler glass fiber filter.

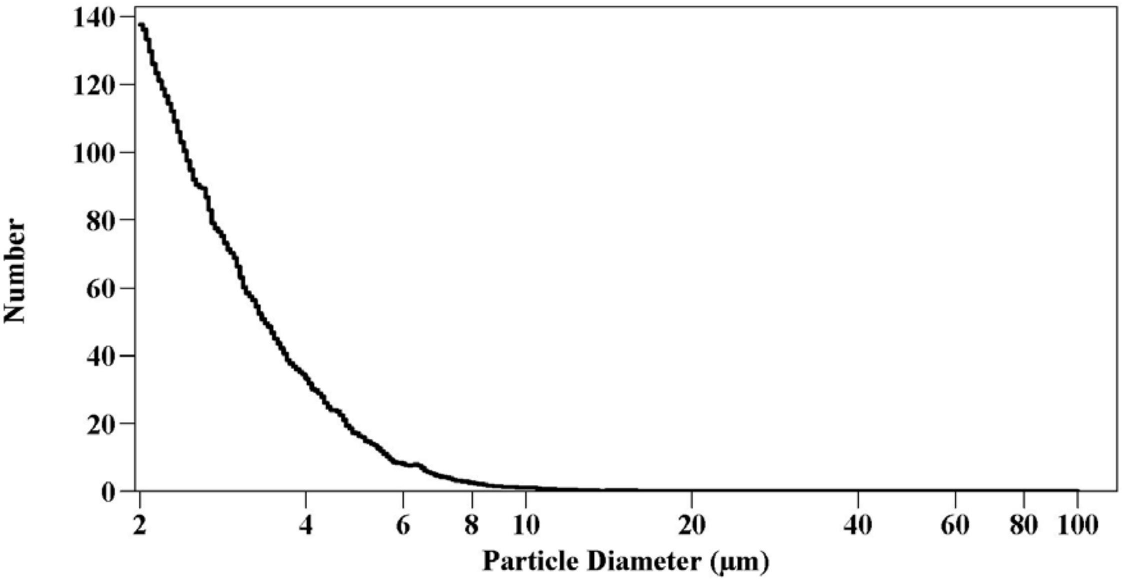

**Figure 75.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank EPA Method 201a, $PM_{10}$ stack sampler glass fiber filter.

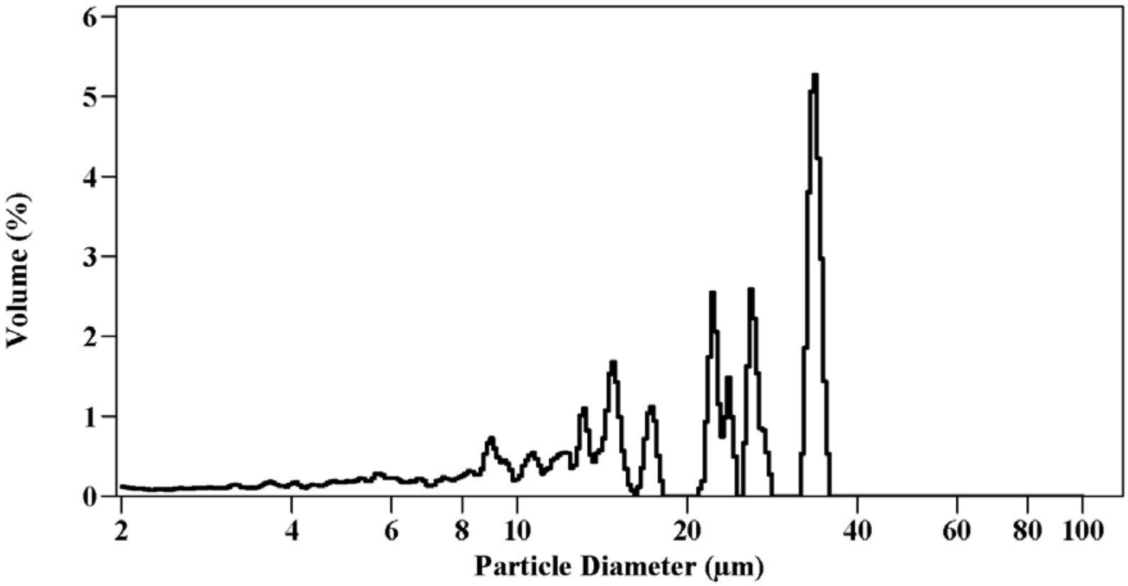

**Figure 76.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank poly-web filter.

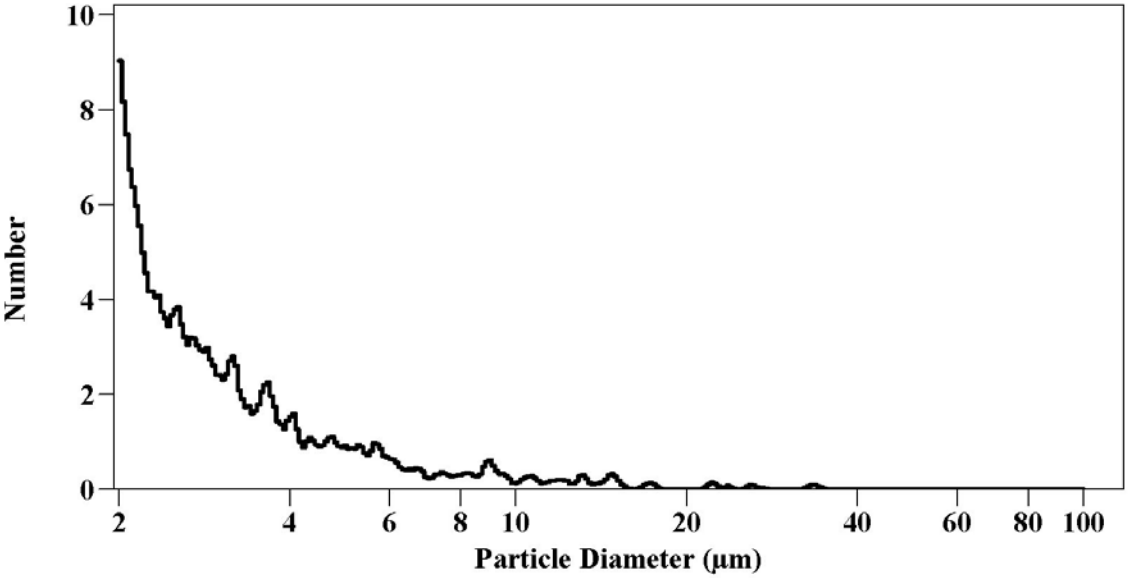

**Figure 77.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank poly-web filter.

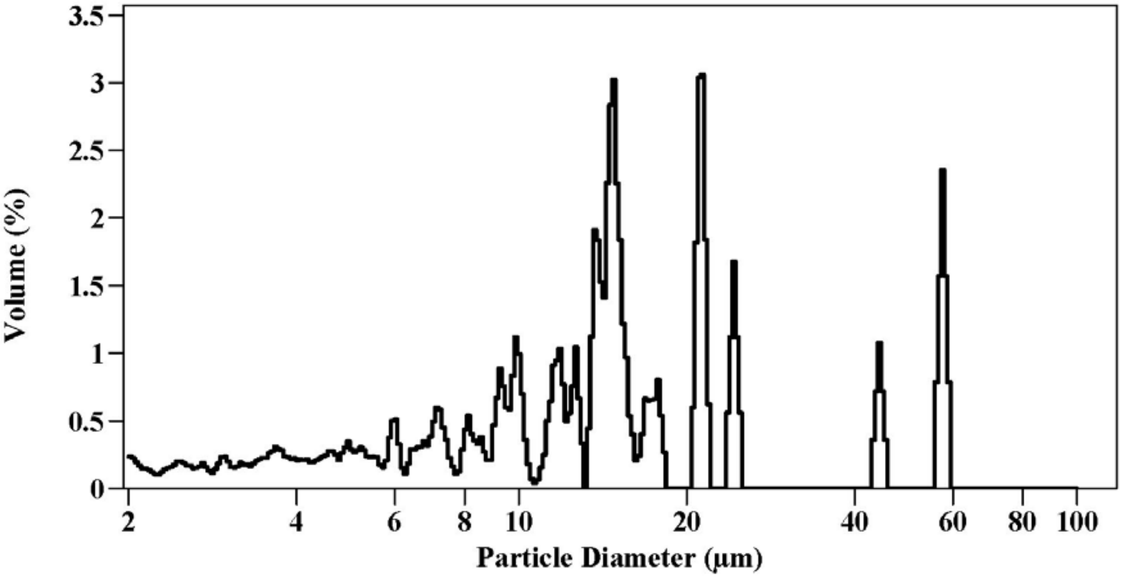

**Figure 78.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank 2 μm Teflon filter.

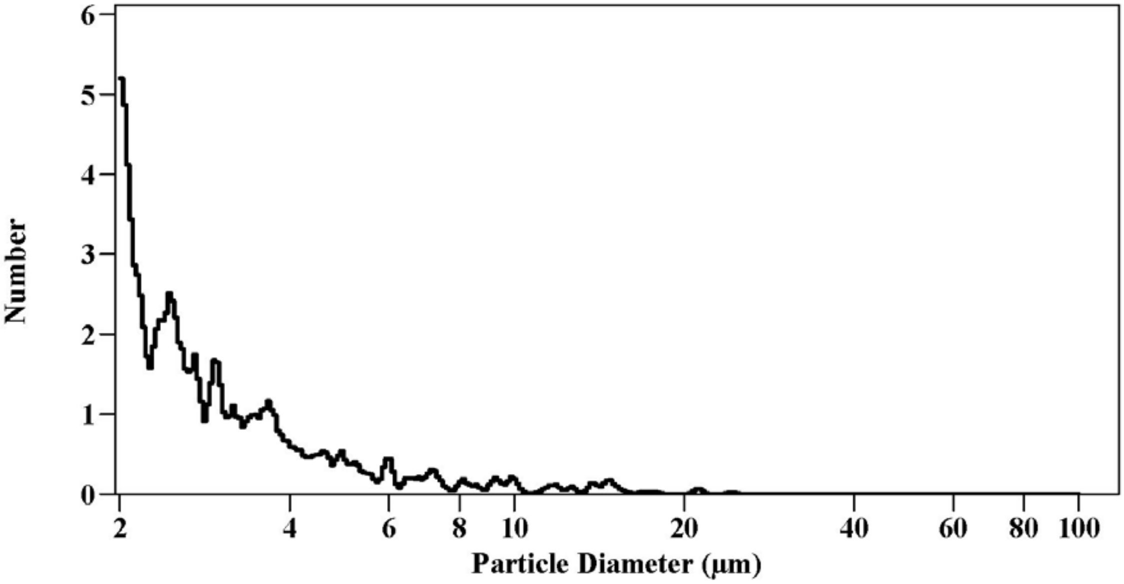

**Figure 79.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank 2 μm Teflon filter.

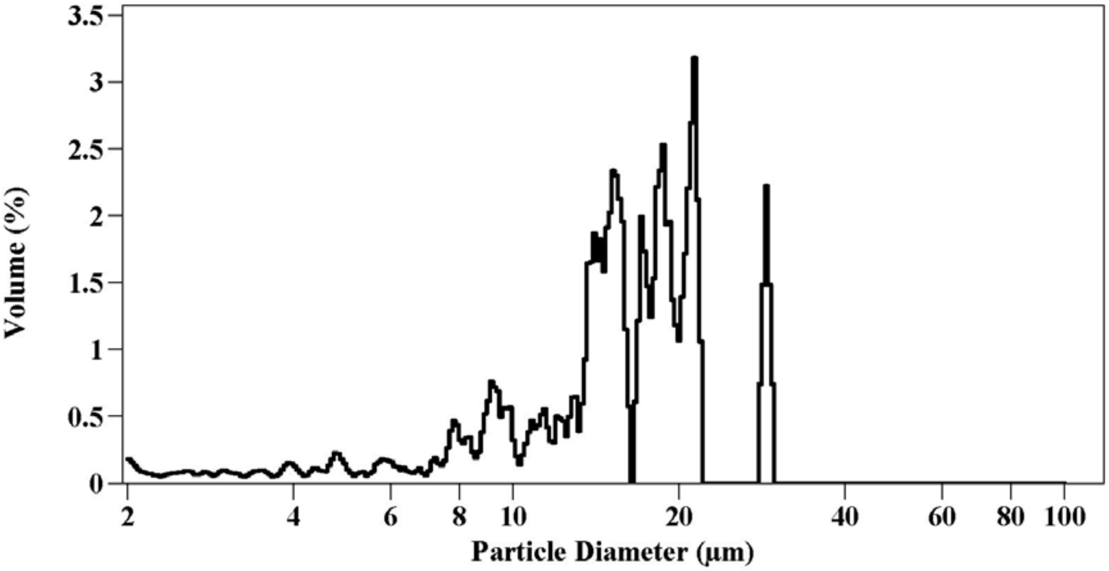

**Figure 80.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank 0.45 μm Teflon filter.

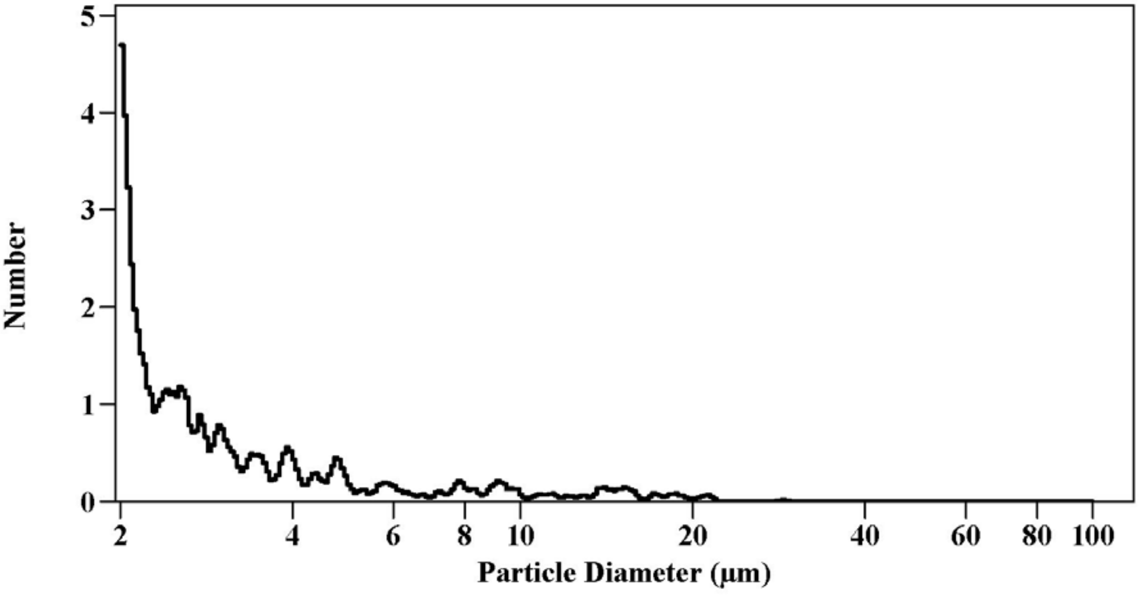

**Figure 81.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank 0.45 μm Teflon filter.

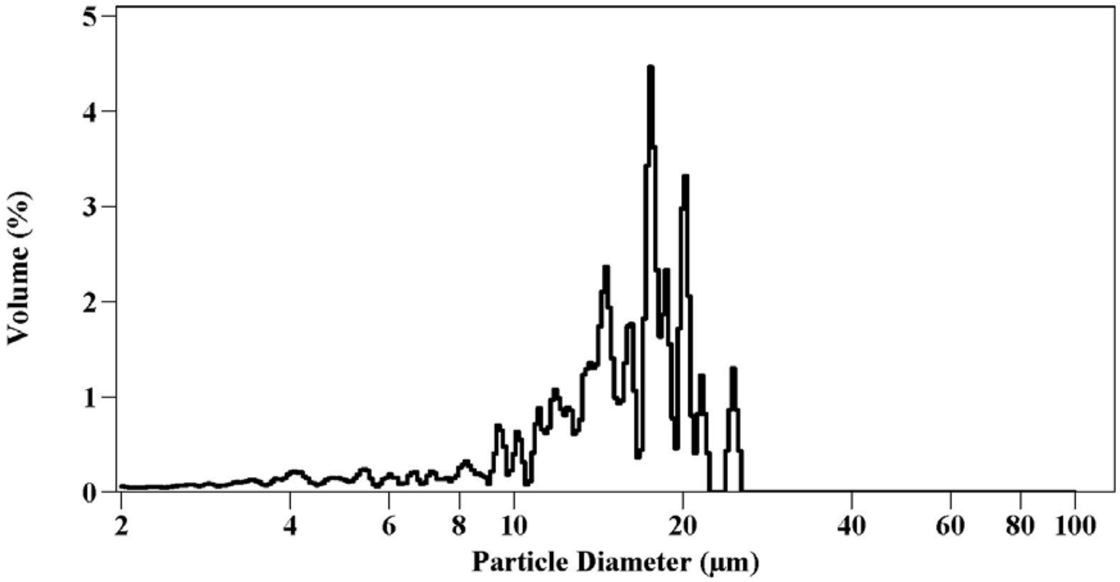

**Figure 82.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank 0.2 µm Teflon filter.

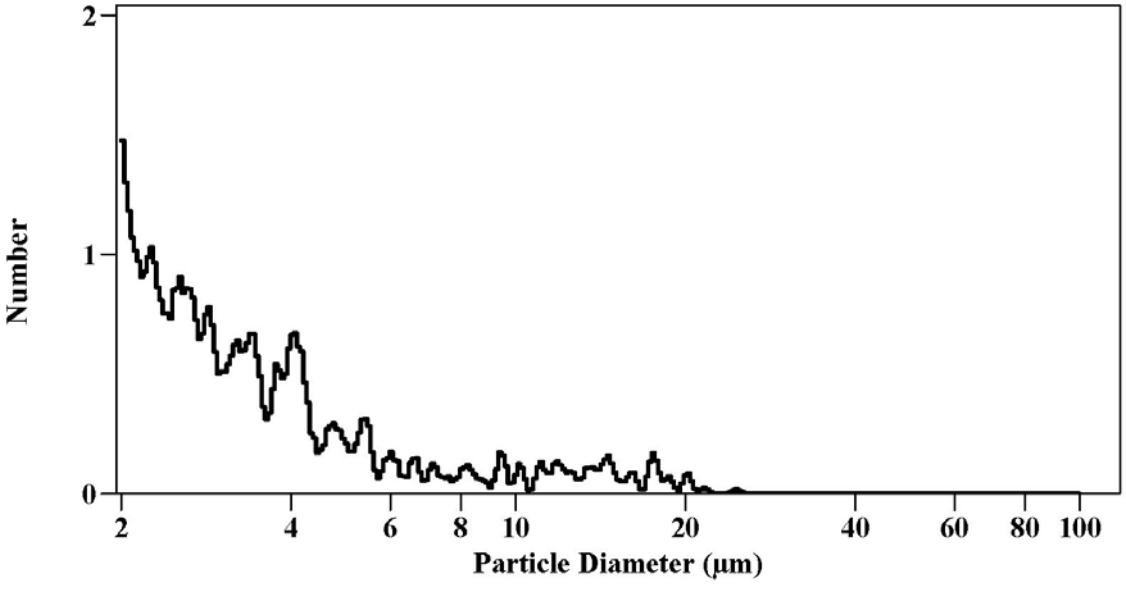

**Figure 83.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a blank 0.2 µm Teflon filter.

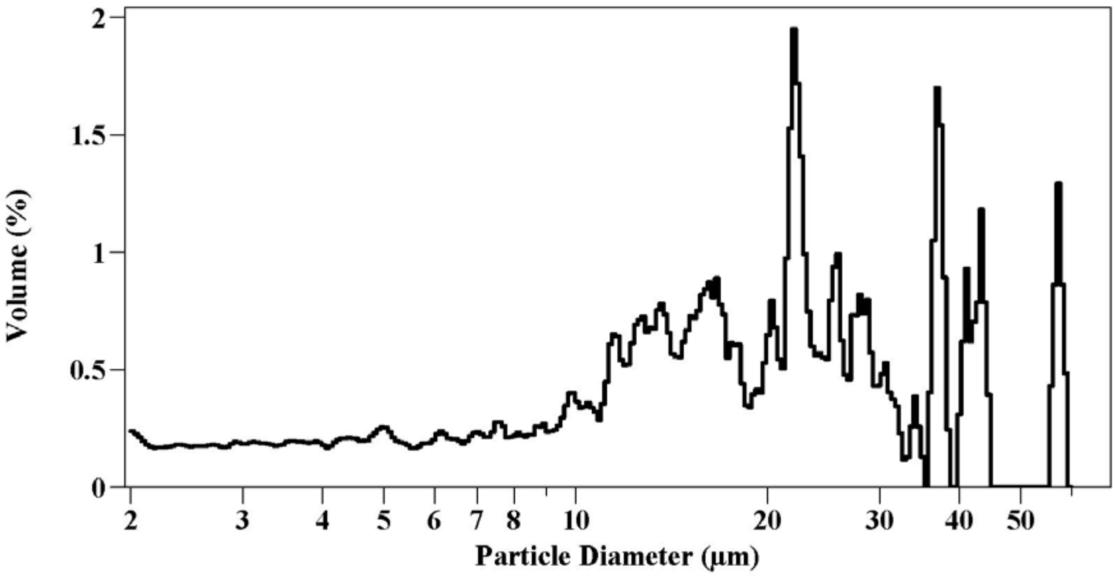

**Figure 84.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a blank polyvinyl-chloride (vertical elutriator sampler) filter.

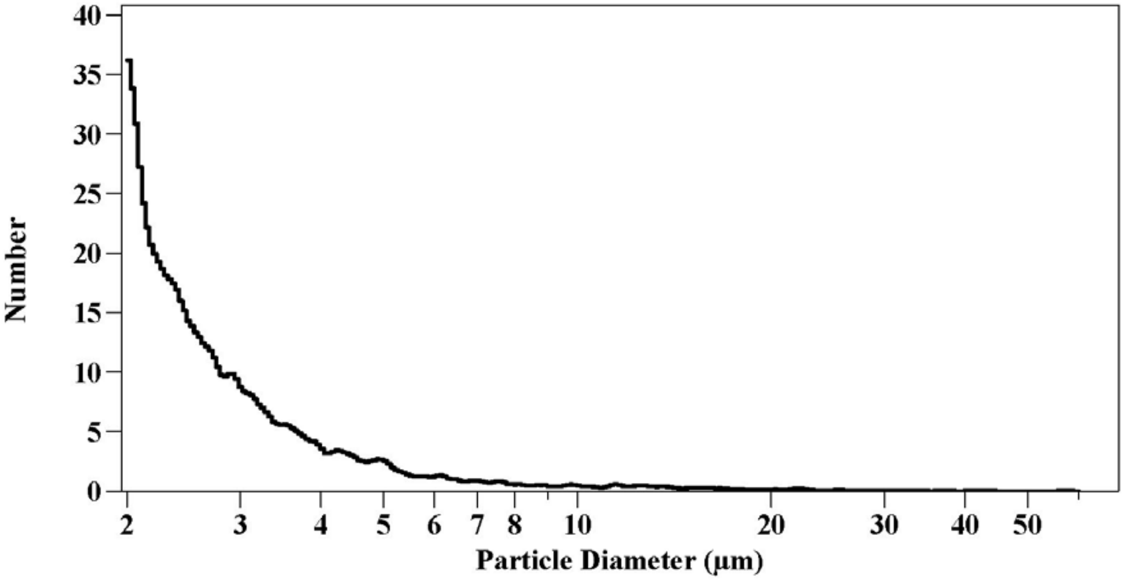

**Figure 85.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a blank polyvinyl-chloride (vertical elutriator sampler) filter.

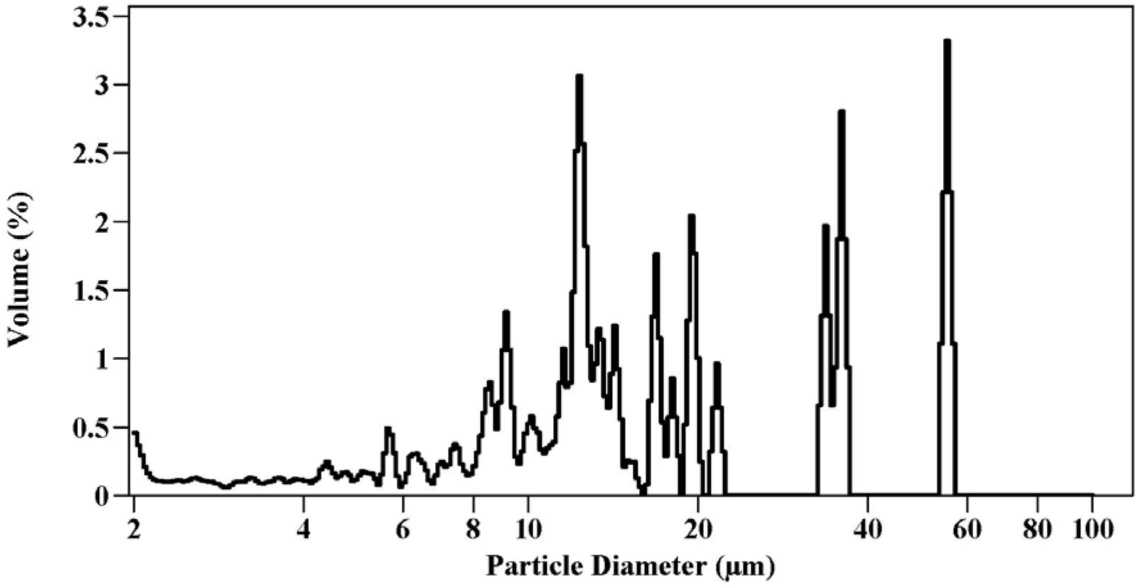

**Figure 86.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a blank cotton swab.

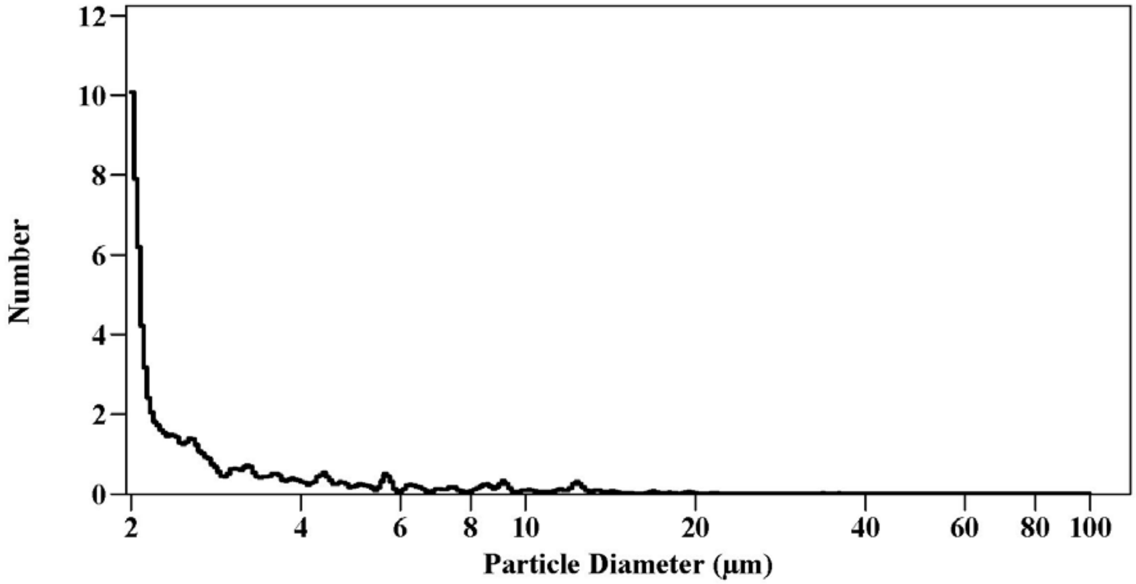

**Figure 87.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a blank cotton swab.

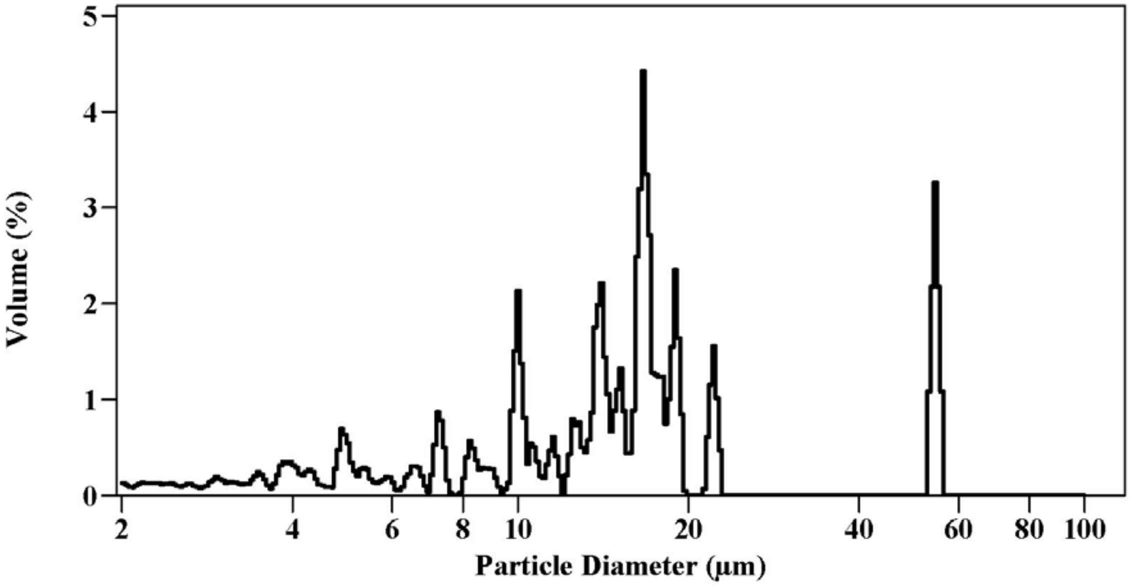

**Figure 88.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a blank foam swab.

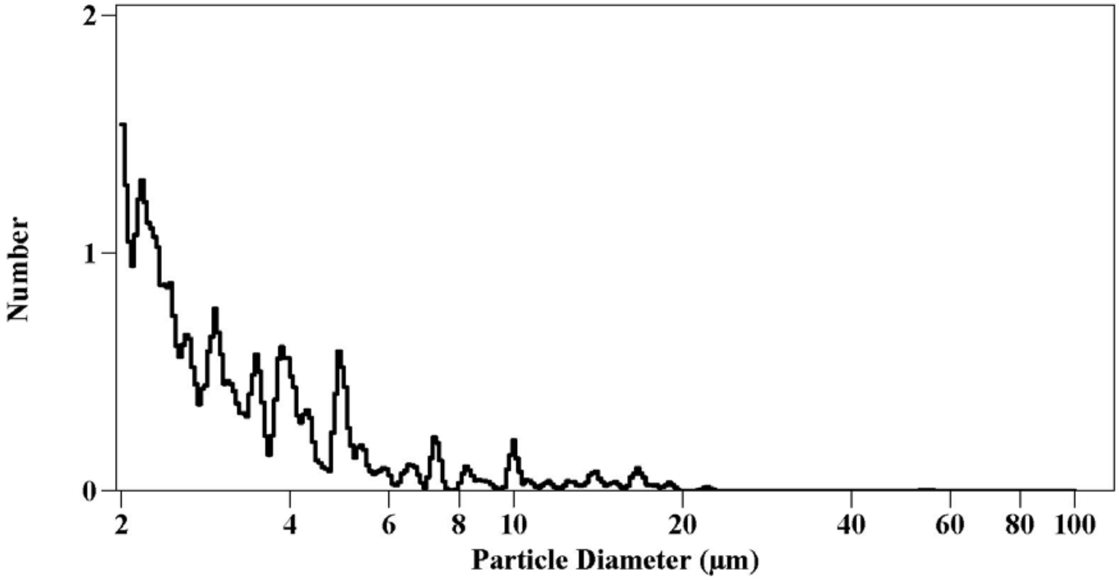

**Figure 89.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a blank foam swab.

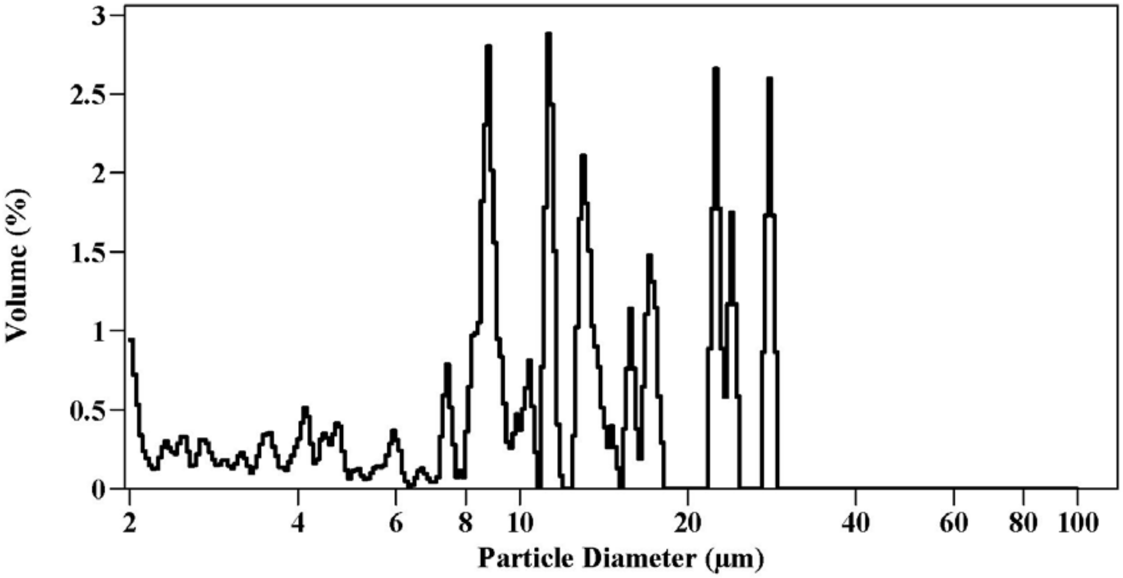

**Figure 90.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a blank nylon swab.

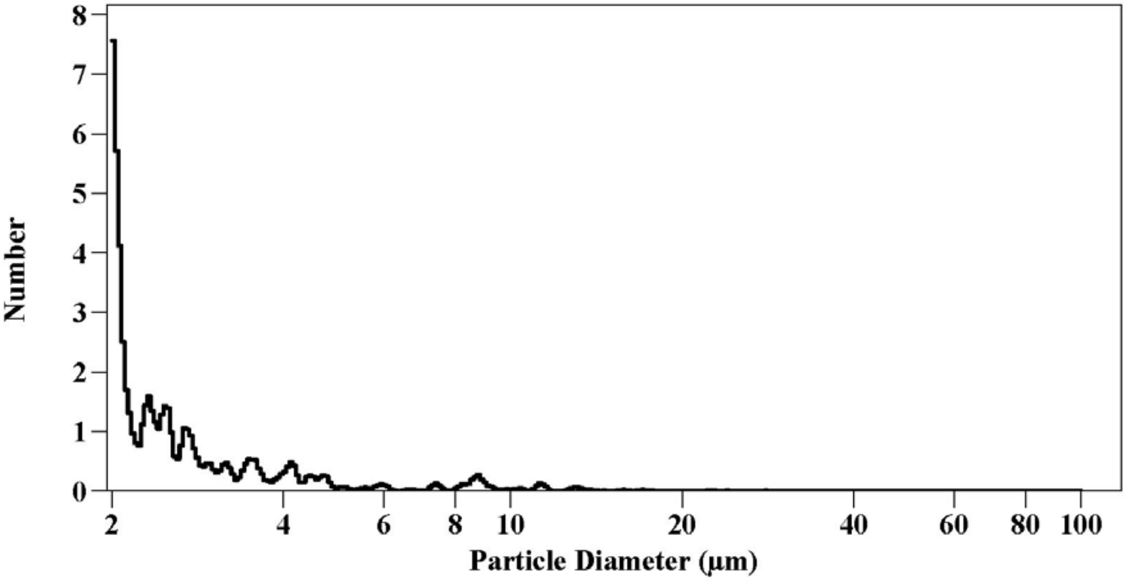

**Figure 91.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a blank nylon swab.

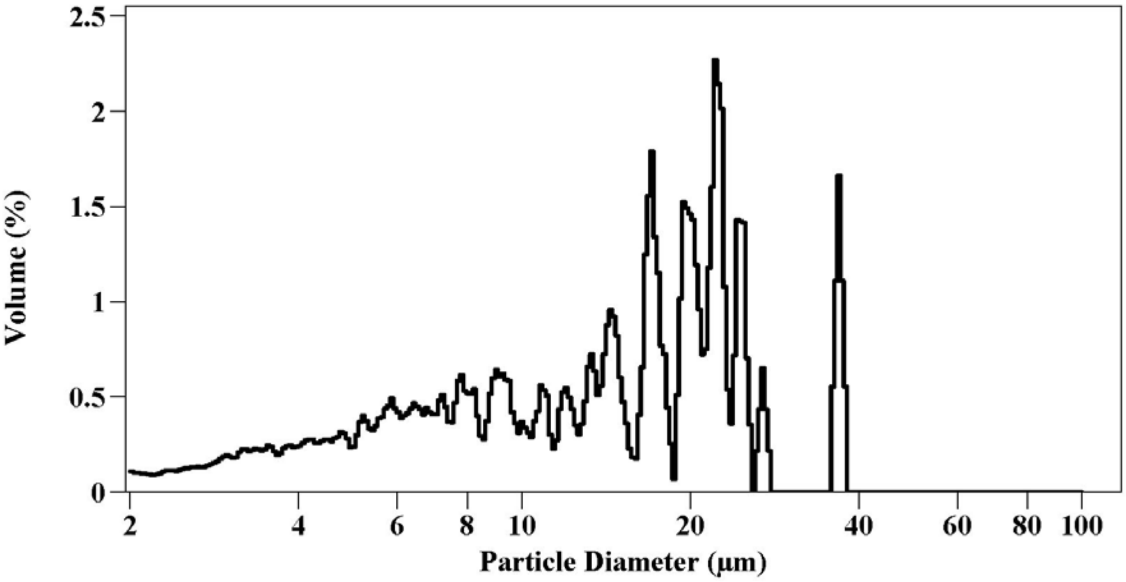

**Figure 92.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a blank TSP high-volume sampler glass fiber filter.

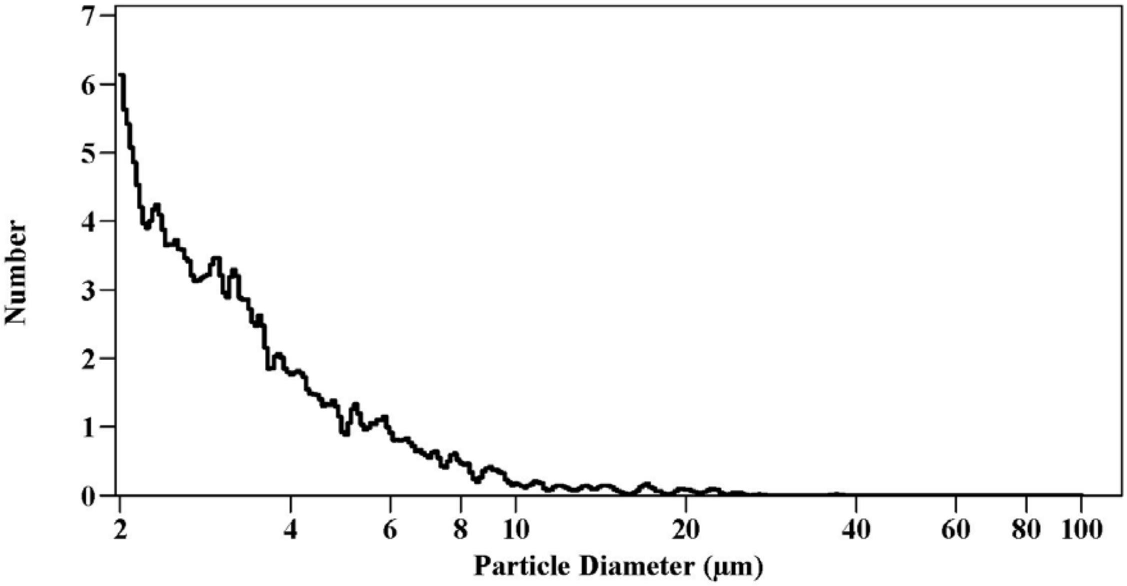

**Figure 93.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a blank TSP high-volume sampler glass fiber filter.

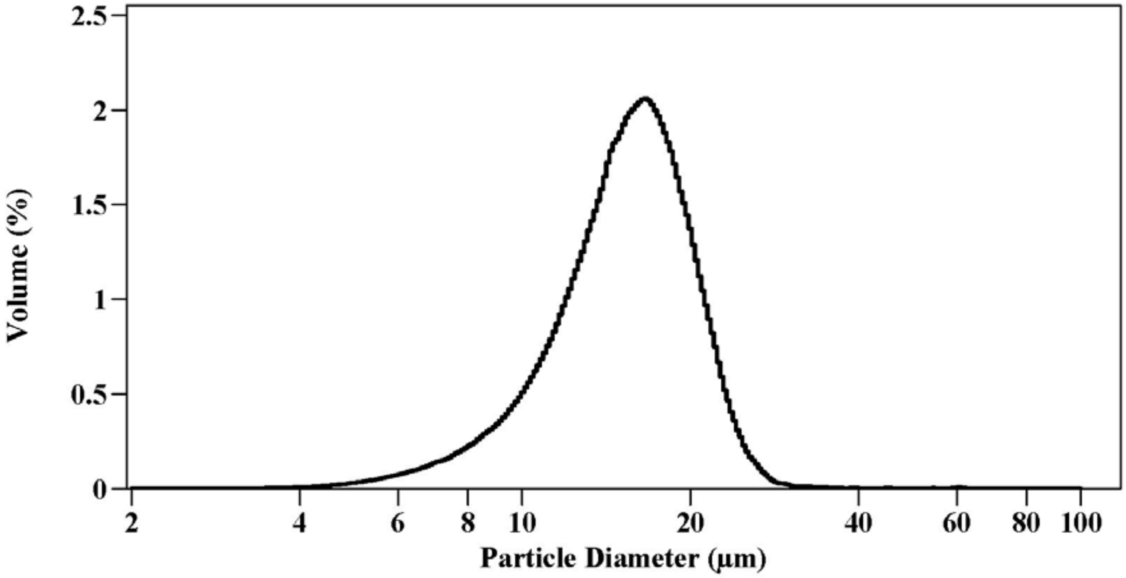

**Figure 94.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for cornstarch.

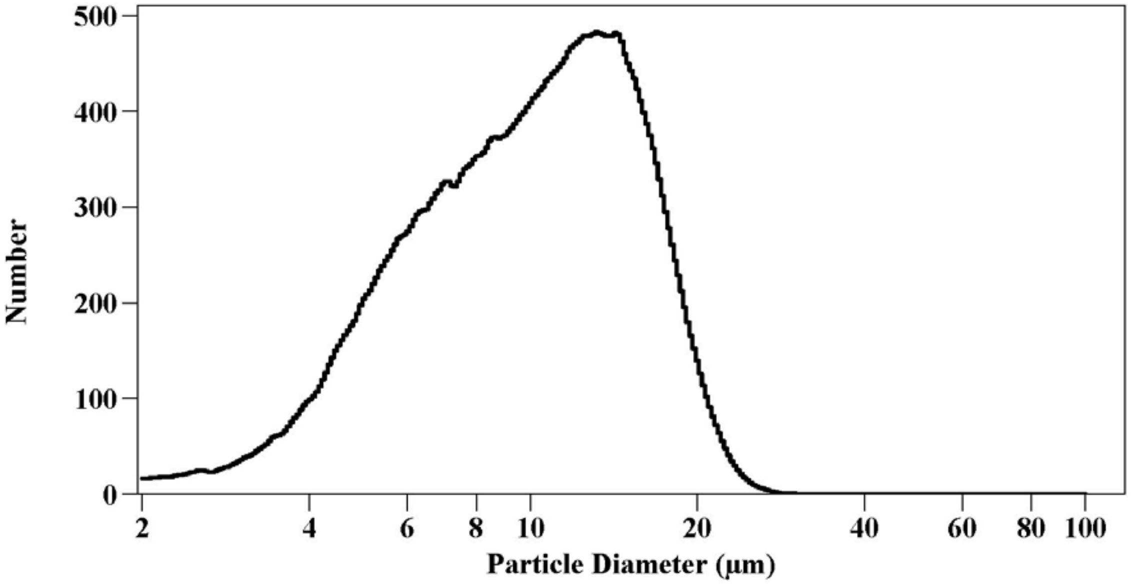

**Figure 95.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for cornstarch.

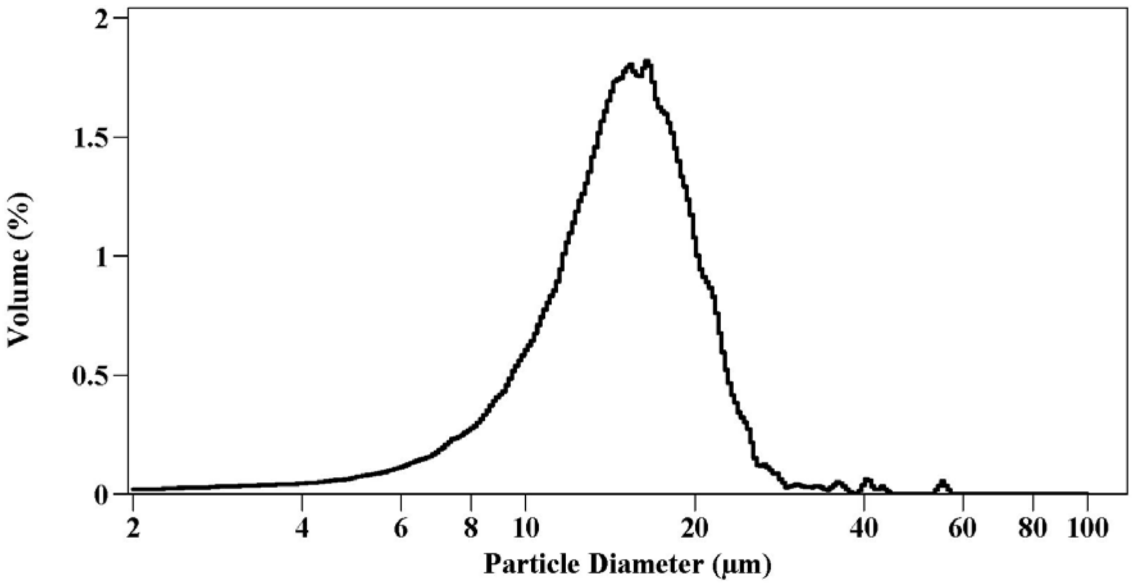

**Figure 96.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾” diameter cut sub-sample from a cornstarch loaded TSP high-volume sampler glass fiber filter.

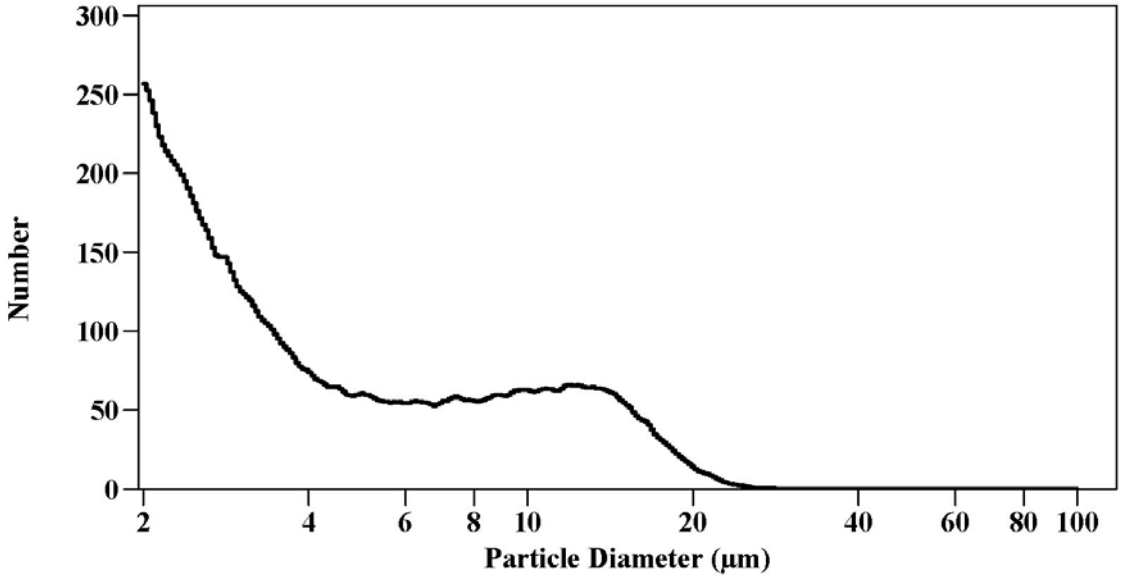

**Figure 97.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾” diameter cut sub-sample from a cornstarch loaded TSP high-volume sampler glass fiber filter.

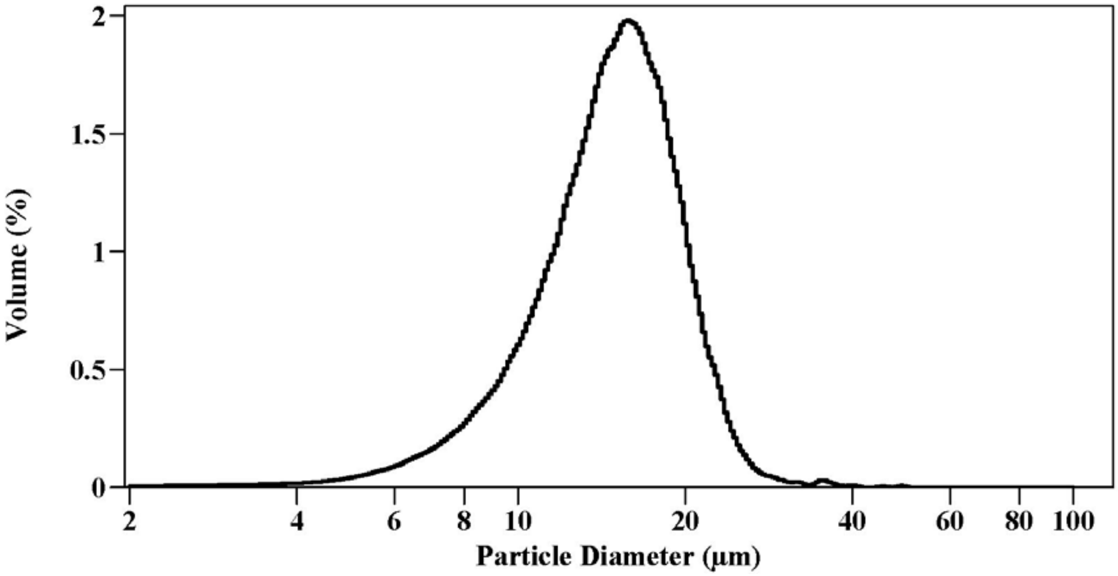

**Figure 98.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a cornstarch loaded TSP high-volume sampler glass fiber filter with a blank TSP high-volume sampler glass fiber filter PSD used as a background and subtracted.

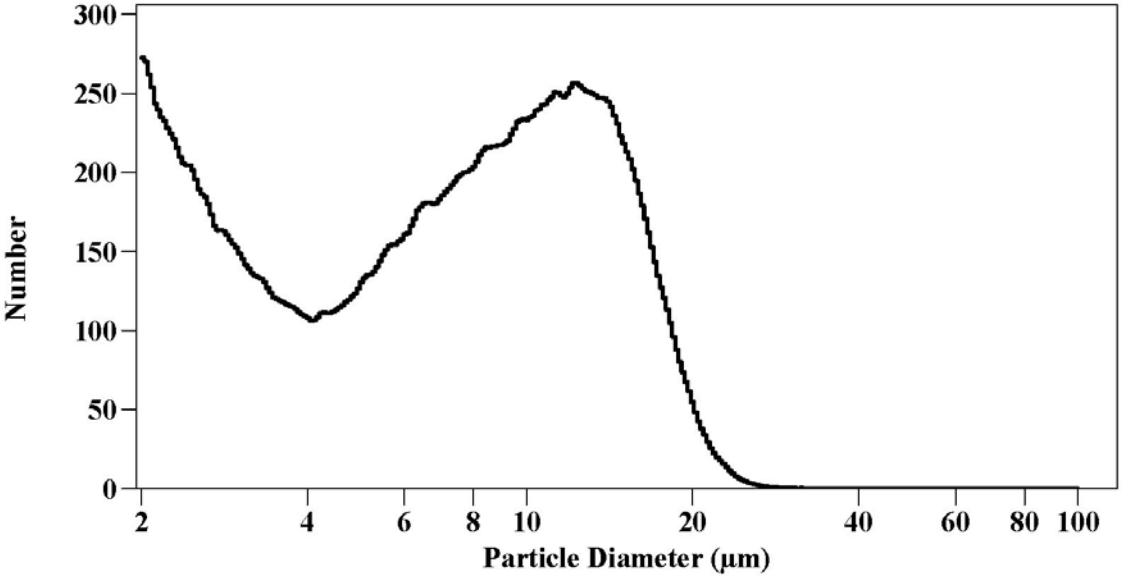

**Figure 99.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for a ¾" diameter cut sub-sample from a cornstarch loaded TSP high-volume sampler glass fiber filter with a blank TSP high-volume sampler glass fiber filter PSD used as a background and subtracted.

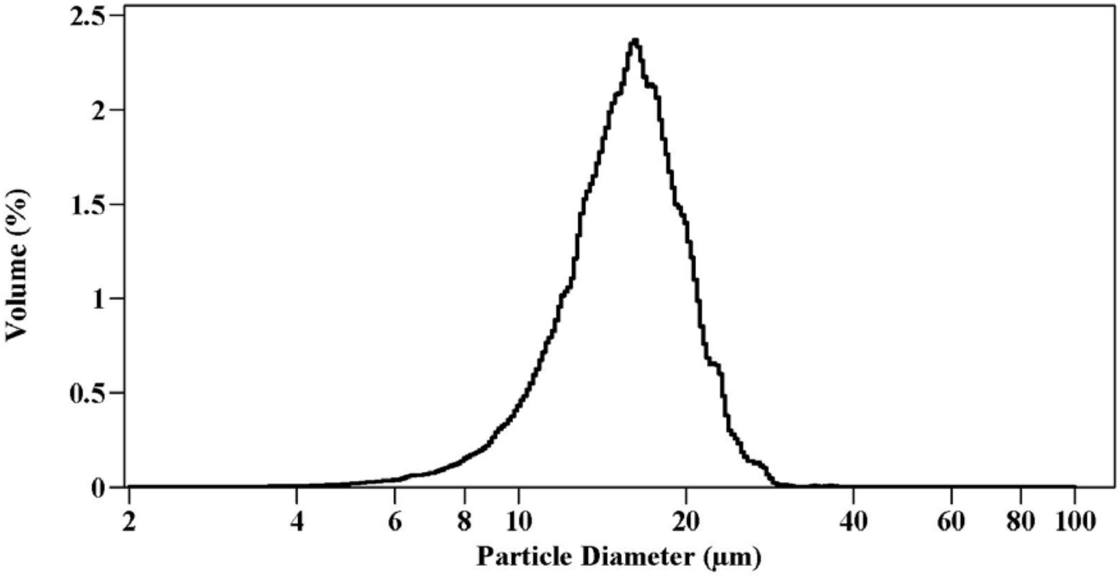

**Figure 100.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a cornstarch loaded TSP high-volume sampler glass fiber filter.

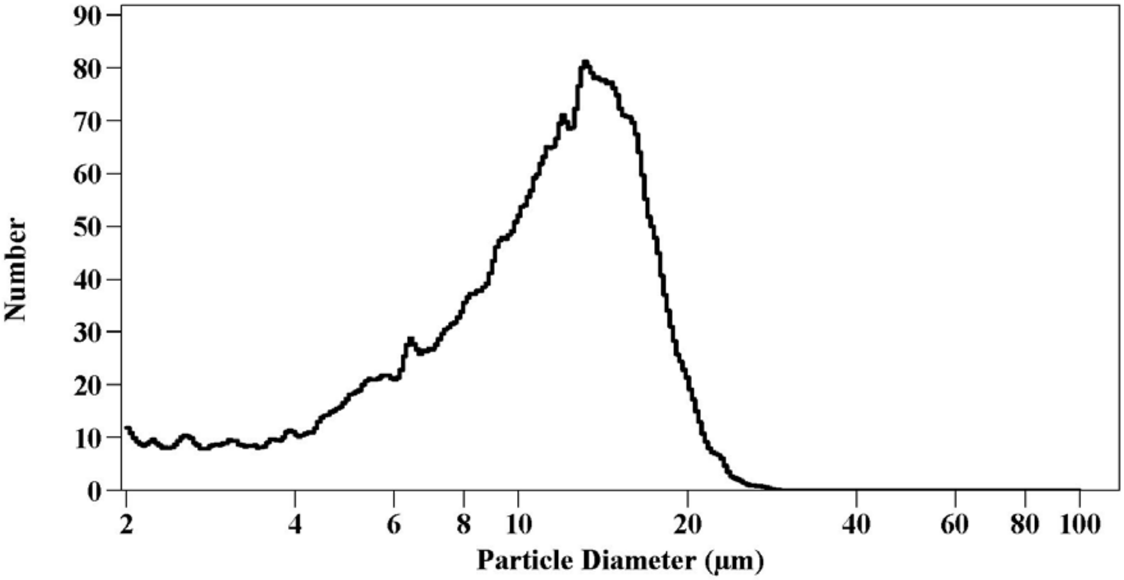

**Figure 101.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a cornstarch loaded TSP high-volume sampler glass fiber filter.

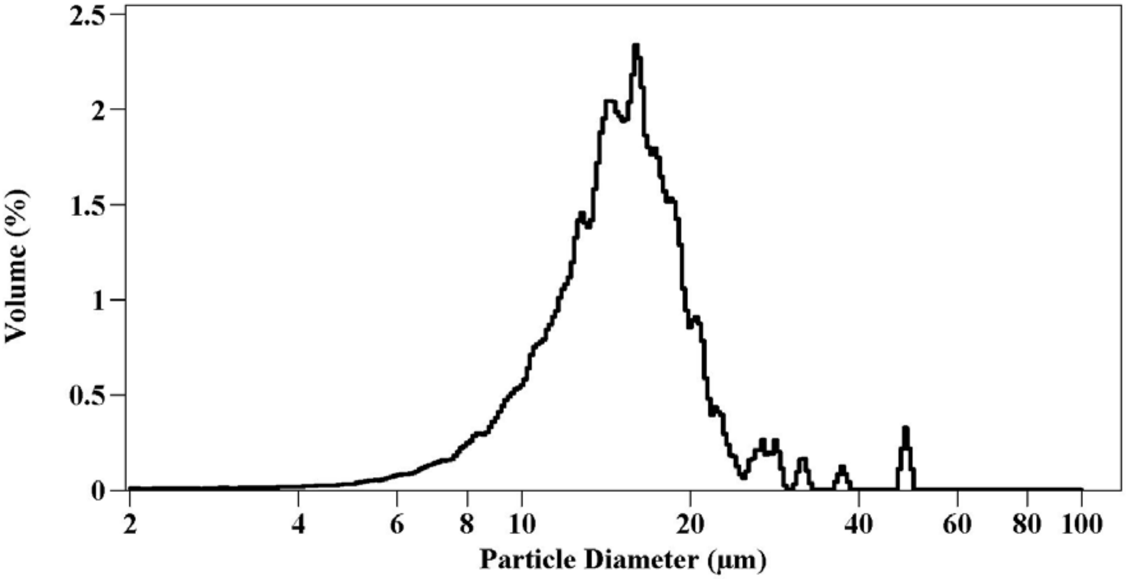

**Figure 102.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a poly-web filter loaded with cornstarch that was transferred from a cornstarch loaded TSP high-volume sampler glass fiber filter.

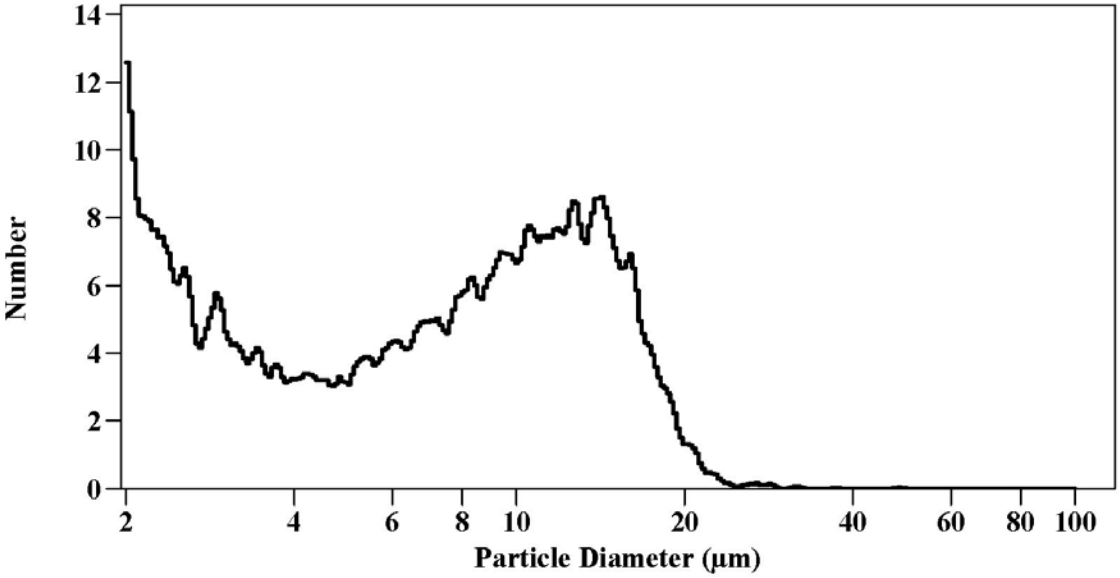

**Figure 103.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a poly-web filter loaded with cornstarch that was transferred from a cornstarch loaded TSP high-volume sampler glass fiber filter.

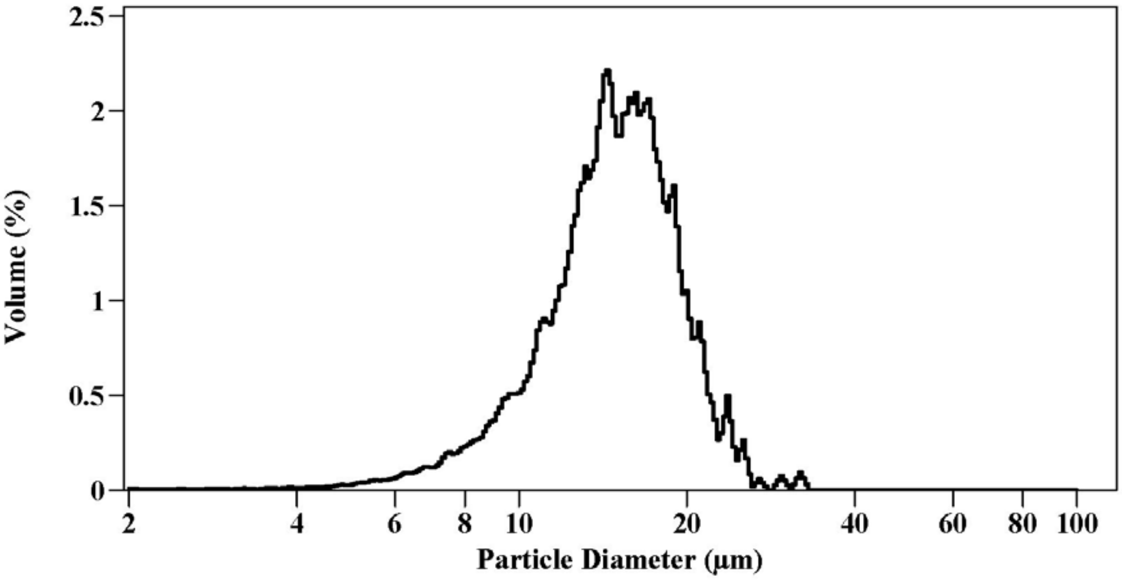

**Figure 104.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a 2 µm Teflon filter loaded with cornstarch that was transferred from a cornstarch loaded TSP high-volume sampler glass fiber filter.

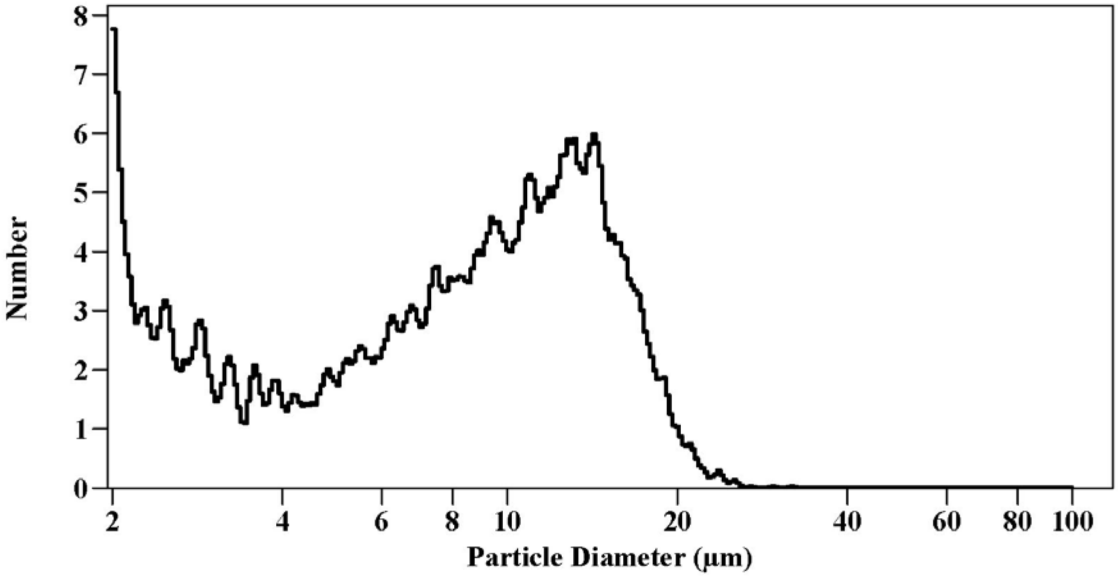

**Figure 105.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a 2 µm Teflon filter loaded with cornstarch that was transferred from a cornstarch loaded TSP high-volume sampler glass fiber filter.

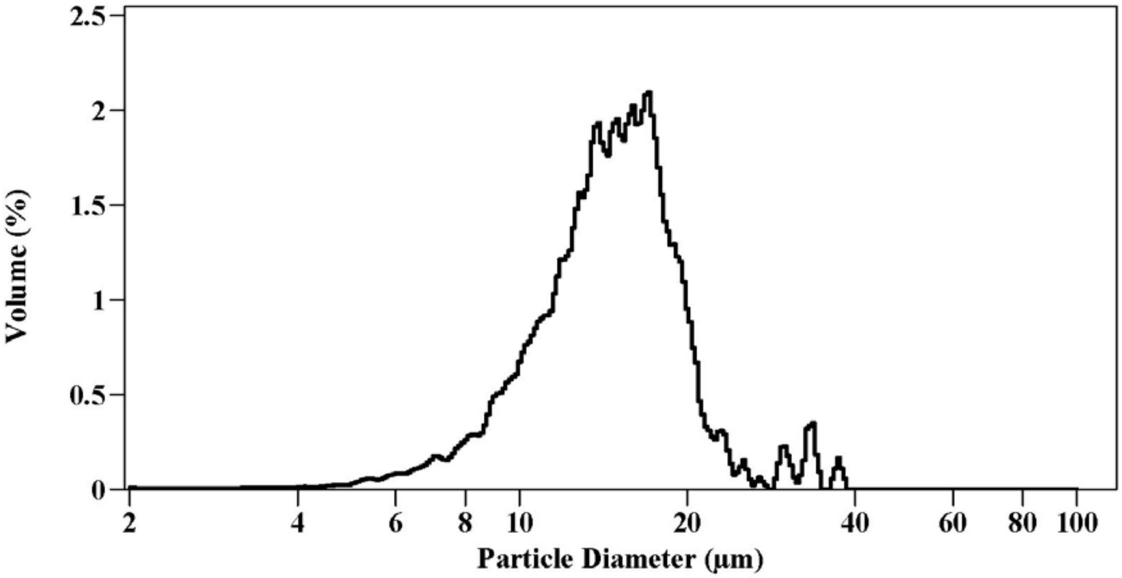

**Figure 106.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a 0.2 µm Teflon filter loaded with cornstarch that was transferred from a cornstarch loaded TSP high-volume sampler glass fiber filter.

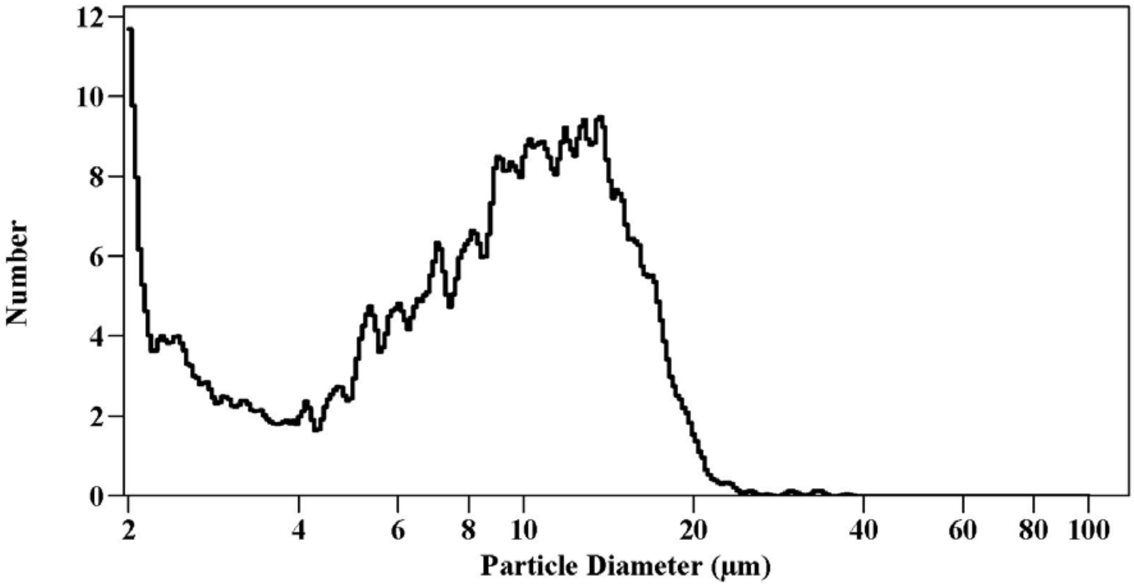

**Figure 107.** Number based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a 0.2 µm Teflon filter loaded with cornstarch that was transferred from a cornstarch loaded TSP high-volume sampler glass fiber filter.

# APPENDIX D

# EVALUATION OF TIME EFFECTS ON PARTICLE SIZE DISTRIBUTION CHARACTERISTICS FOR PM DISPERSED IN ELECTROLYTE

A study was conducted to determine how cornstarch dispersed in electrolyte behaved after remaining in the electrolyte for various time frames. The treatments included: 1) subjecting the cornstarch and electrolyte solution to an ultrasonic bath for 5 minutes before performing a PSD analysis; 2) subjecting the solution to an ultrasonic bath for 15 minutes before performing a PSD analysis; 3) subjecting a solution to a 15 minute ultrasonic bath, allowing the solution to set for 4 days and subjecting the solution to a 15 minute ultrasonic bath before running a PSD analysis; 4) the process described is item 3 was completed for 6, 12, 16, 20, and 33 days. Three replications and three repeated measures were completed for each treatment. The PROC Mixed (Littell, et al., 1996) procedure in SAS was utilized to compare PSD characteristics. Results from the study are listed in Table 34. Figures 108 through 115 show the average volume based PSD associated with each treatment. Several significant differences were detected between the treatments. In general, the MMD decreased and the GSD increased with an increase in the time the cornstarch remained in the solution. Although there were significant differences detected between storing the solution from 6 to 33 days, the differences are relatively small. The largest differences in MMD were observed when comparing the 5 and 15 minute ultrasonic bath treatments. The MMD decreased by about 2 μm when comparing the 5 to 15 minute ultrasonic bath treatments.

**Table 34.** Particle size distribution characteristics associated with PM dispersed and stored in electrolyte for various time periods.

| Time Frame | MMD (μm) | $D_{15.9}$ (μm) | $D_{84.1}$ (μm) | % less than 10 μm | GSD |
|---|---|---|---|---|---|
| 5-minute[a] | 18.7 a | 10.3 a | 27.1 a | 14.9 g | 1.62 ef |
| 15-minute[a] | 16.8 b | 9.4 b | 24.7 b | 18.3 f | 1.62 f |
| 4 days[b] | 15.6 c | 8.9 c | 23.4 d | 20.1 e | 1.63 ef |
| 6 days[b] | 14.9 d | 8.5 d | 22.8 e | 23.1 d | 1.64 de |
| 12 days[b] | 14.5 e | 8.3 def | 22.6 e | 24.5 bc | 1.65 cd |
| 16 days[b] | 14.3 f | 8.2 ef | 22.6 e | 25.3 ab | 1.67 bc |
| 20 days[b] | 14.6 e | 8.3 de | 23.8 c | 24.2 c | 1.69 a |
| 33 days[b] | 14.1 f | 8.1 f | 22.7 e | 25.8 a | 1.68 ab |

[a] Ultrasonic bath time frame.

[b] Subject to a 15-minute ultrasonic bath, stored for the stated time frame, and subjected to a 15-minute ultrasonic bath.

Means in the same column followed by the same letter are not statistically different at the 0.05 level of significance.

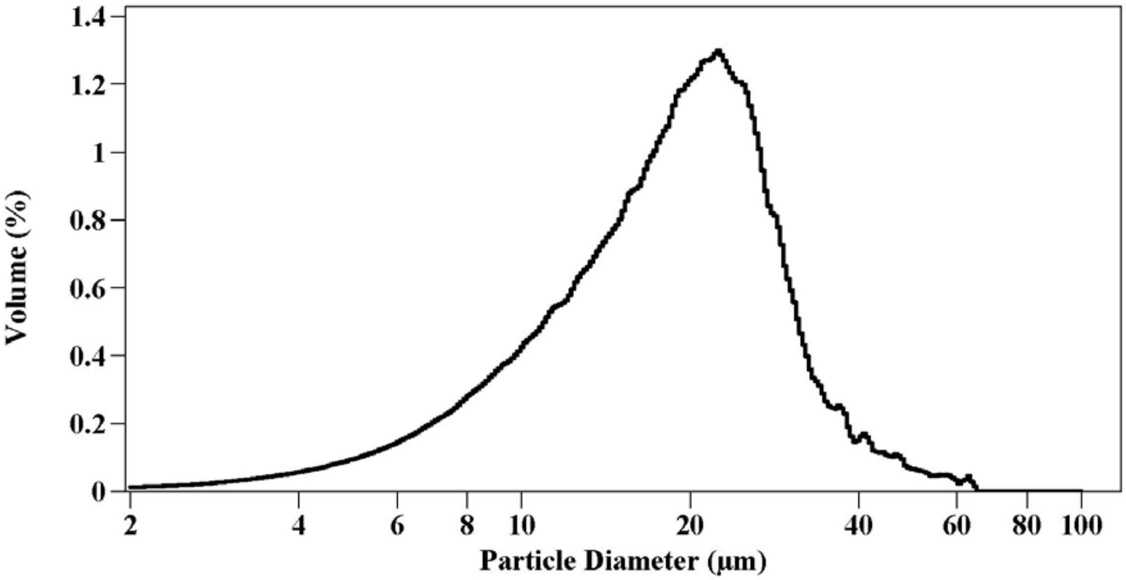

**Figure 108.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for PM captured from a cotton gin exhaust dispersed in an electrolyte solution through the use of an ultrasonic bath for five minutes.

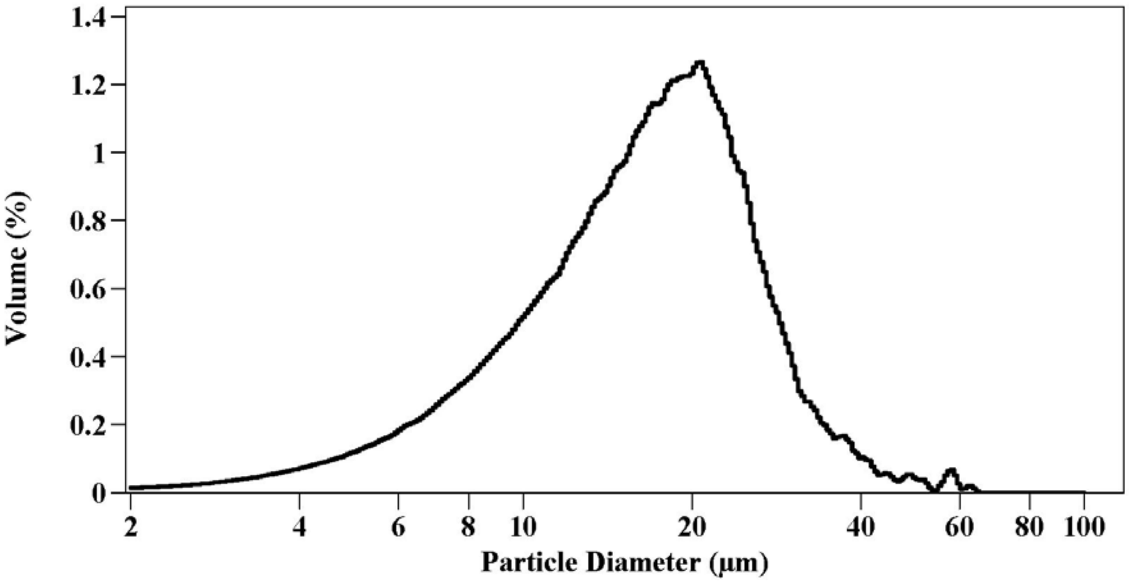

**Figure 109.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for PM captured from a cotton gin exhaust dispersed in an electrolyte solution through the use of an ultrasonic bath for fifteen minutes.

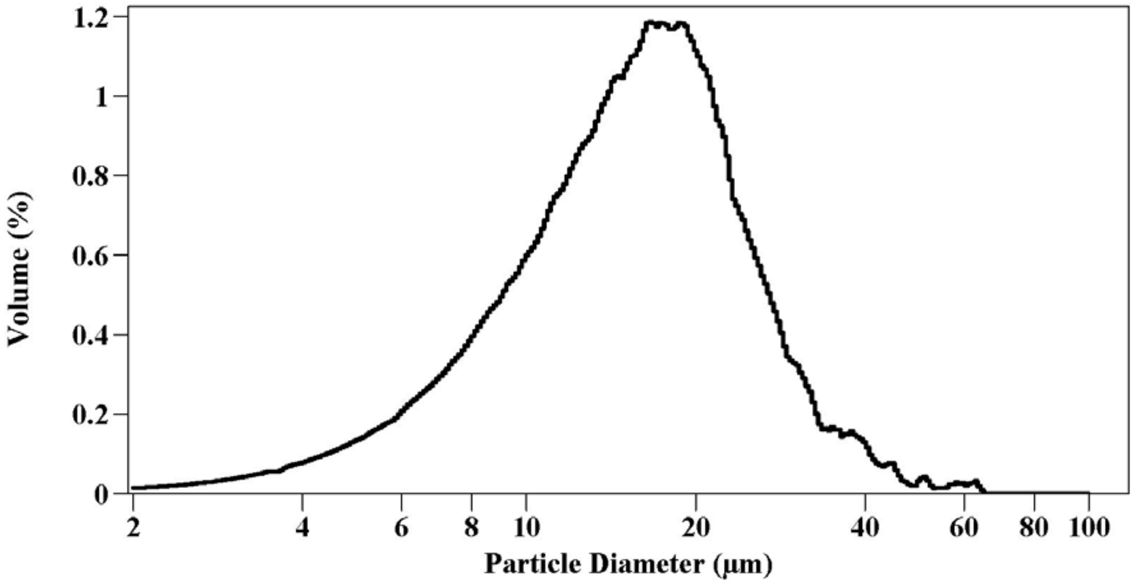

**Figure 110.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for PM captured from a cotton gin exhaust dispersed in an electrolyte solution through the use of an ultrasonic bath for fifteen minutes; the solution allowed to set for four days and then subjected to an additional fifteen-minute ultrasonic bath.

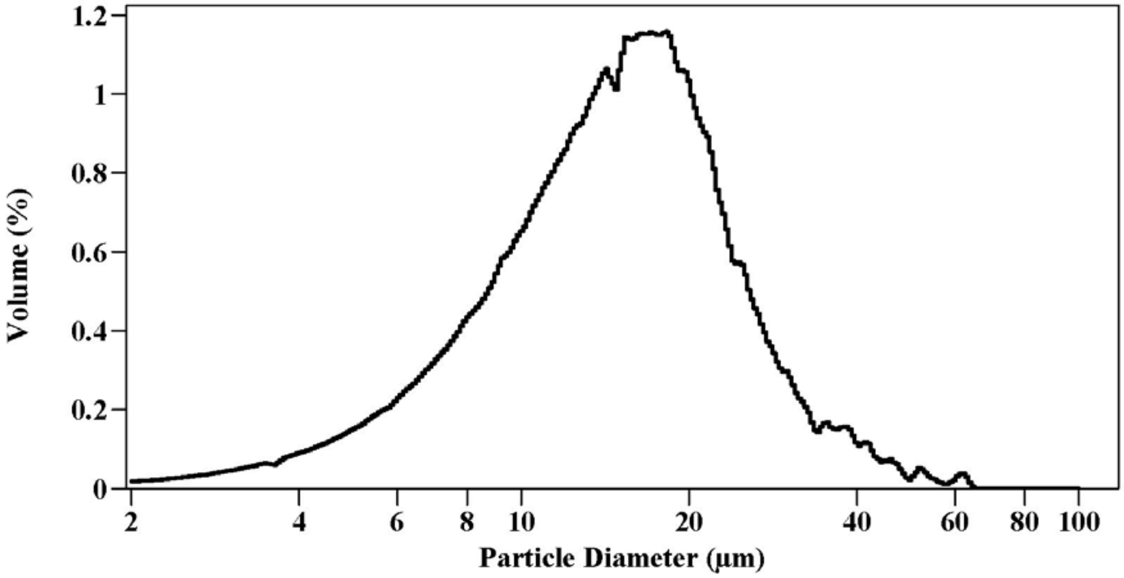

**Figure 111.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for PM captured from a cotton gin exhaust dispersed in an electrolyte solution through the use of an ultrasonic bath for fifteen minutes; the solution allowed to set for six days and then subjected to an additional fifteen-minute ultrasonic bath.

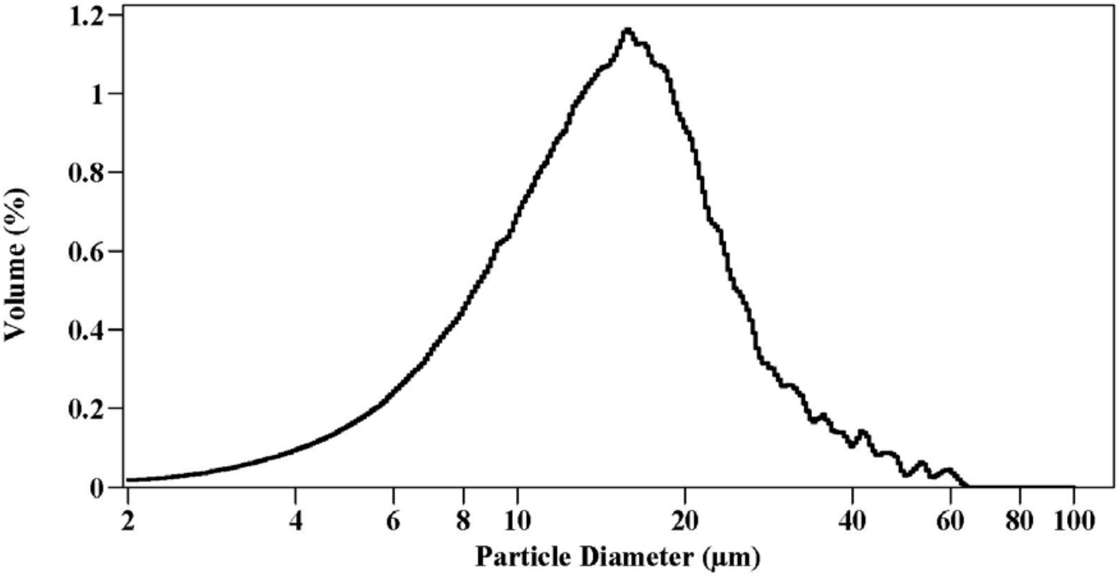

**Figure 112.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for PM captured from a cotton gin exhaust dispersed in an electrolyte solution through the use of an ultrasonic bath for fifteen minutes; the solution allowed to set for twelve days and then subjected to an additional fifteen-minute ultrasonic bath.

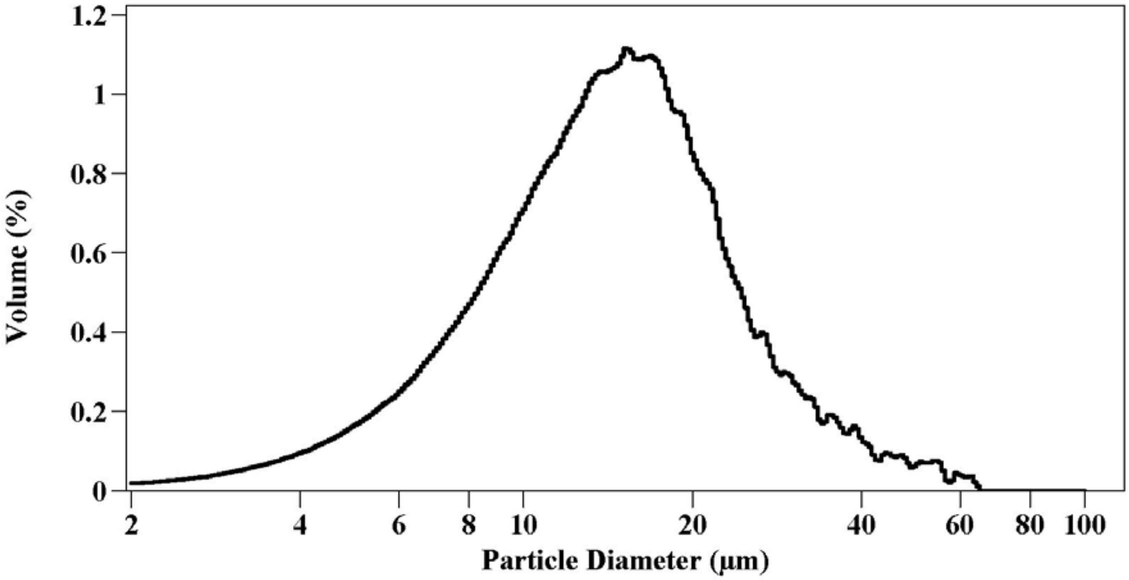

**Figure 113.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for PM captured from a cotton gin exhaust dispersed in an electrolyte solution through the use of an ultrasonic bath for fifteen minutes; the solution allowed to set for sixteen days and then subjected to an additional fifteen-minute ultrasonic bath.

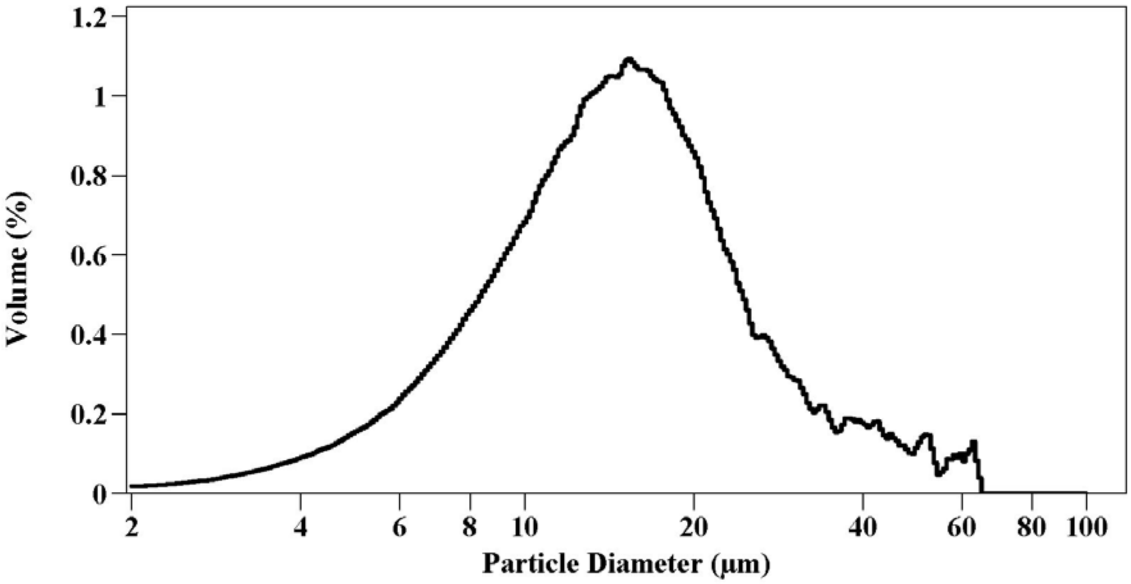

**Figure 114.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for PM captured from a cotton gin exhaust dispersed in an electrolyte solution through the use of an ultrasonic bath for fifteen minutes; the solution allowed to set for twenty days and then subjected to an additional fifteen-minute ultrasonic bath.

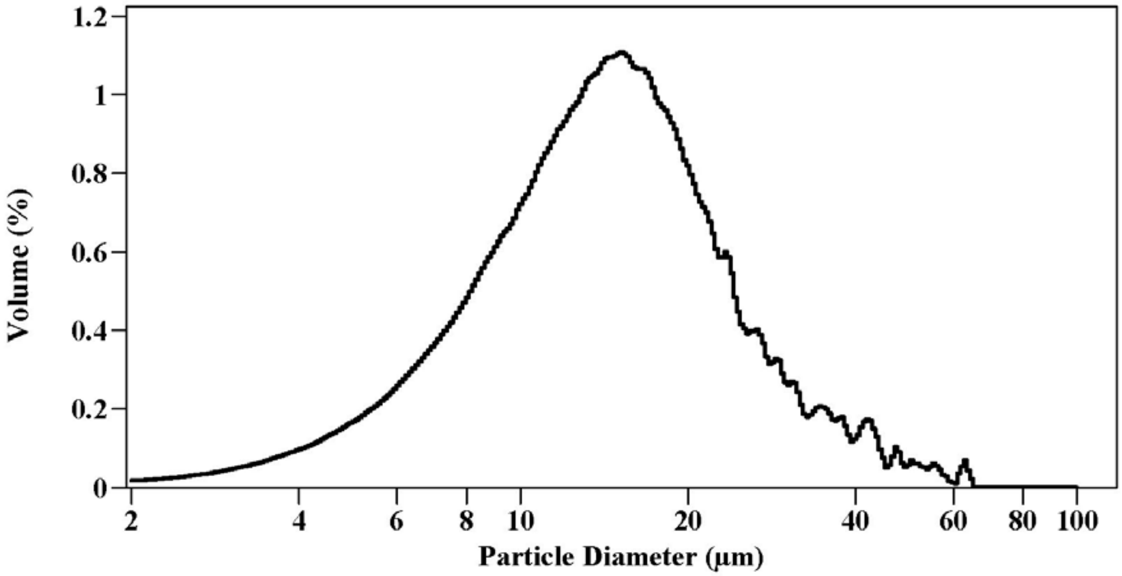

**Figure 115.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) for PM captured from a cotton gin exhaust dispersed in an electrolyte solution through the use of an ultrasonic bath for fifteen minutes; the solution allowed to set for thirty-three days and then subjected to an additional fifteen-minute ultrasonic bath.

# APPENDIX E

# PARTICLE SIZE DISTRIBUTION CHARACTERISTICS OF STRIPPER AND PICKER COTTON GIN TRASH, SEGREGATED BY SIMILAR PROCESS STREAMS, GINNED AT THE USDA-ARS COTTON GINNING RESEARCH UNIT IN STONEVILLE, MS (MICROGIN)

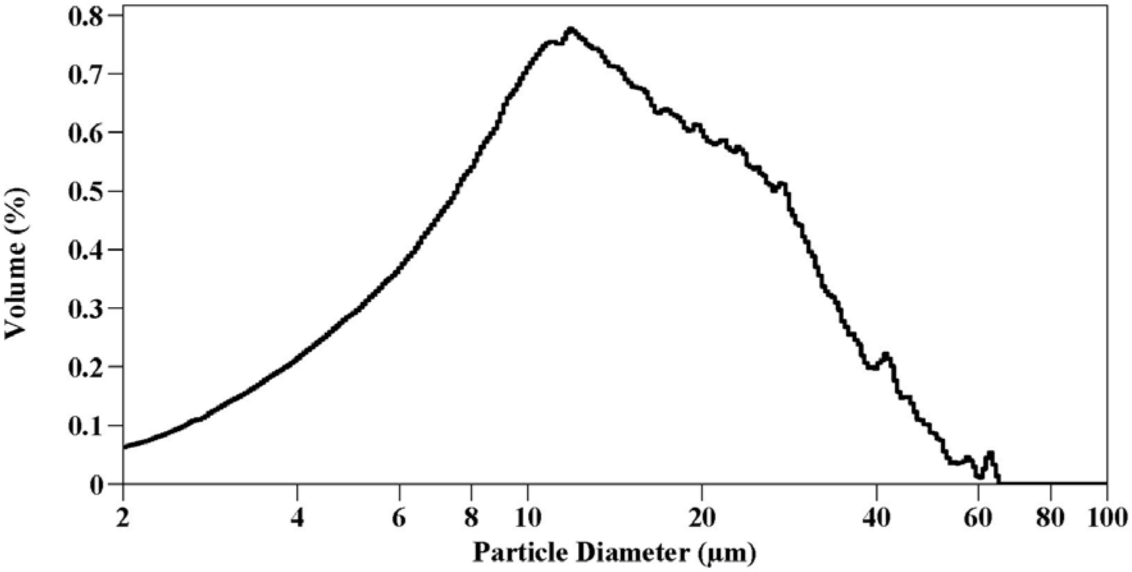

**Figure 116.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed picker gin trash from the first cylinder cleaner.

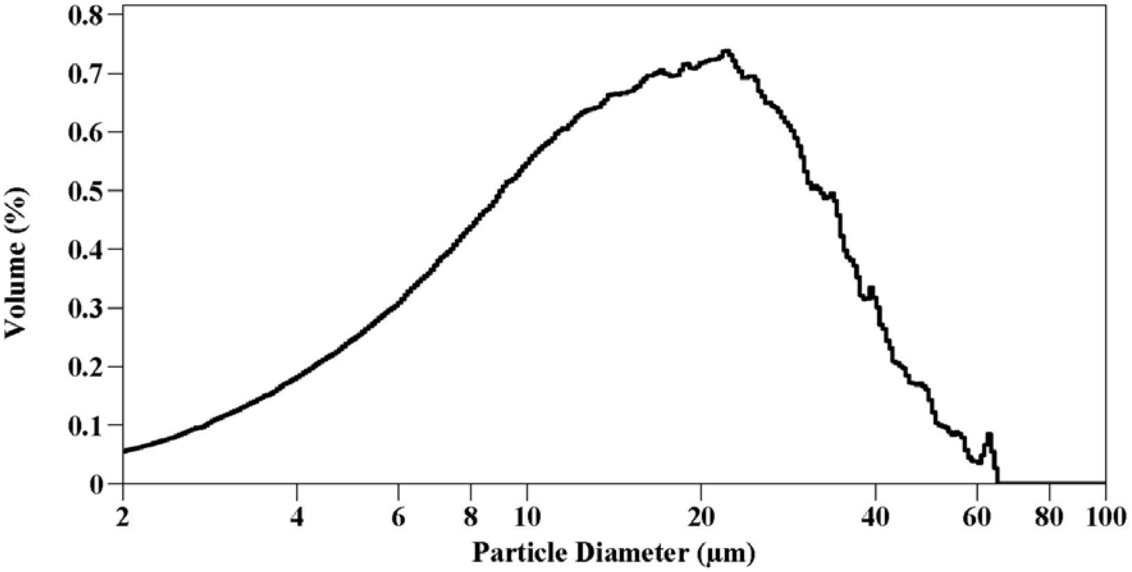

**Figure 117.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed stripper gin trash from the first cylinder cleaner.

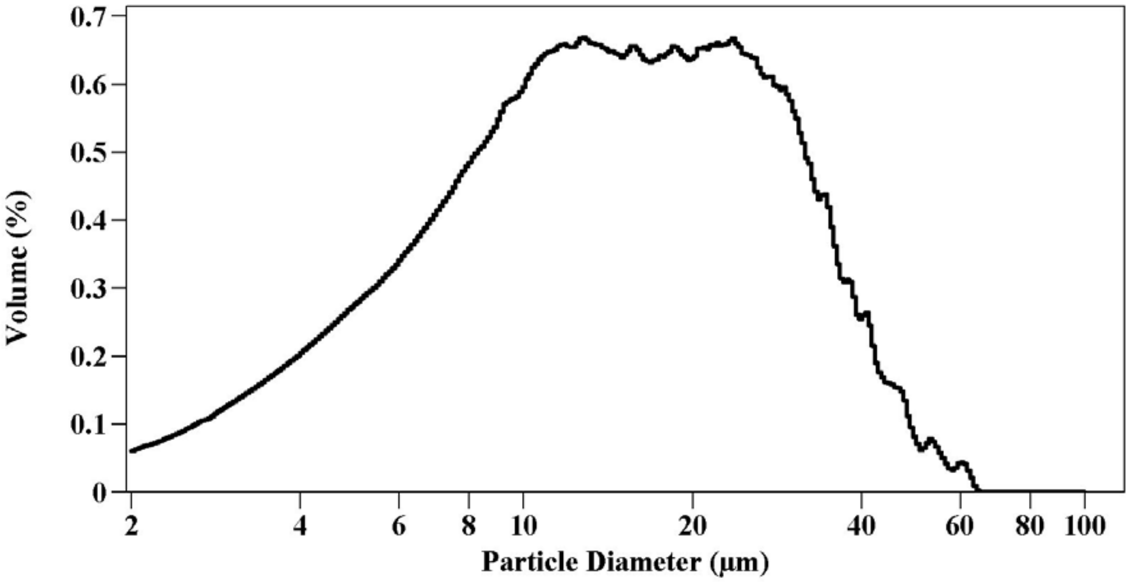

**Figure 118.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed picker gin trash from the first stick machine.

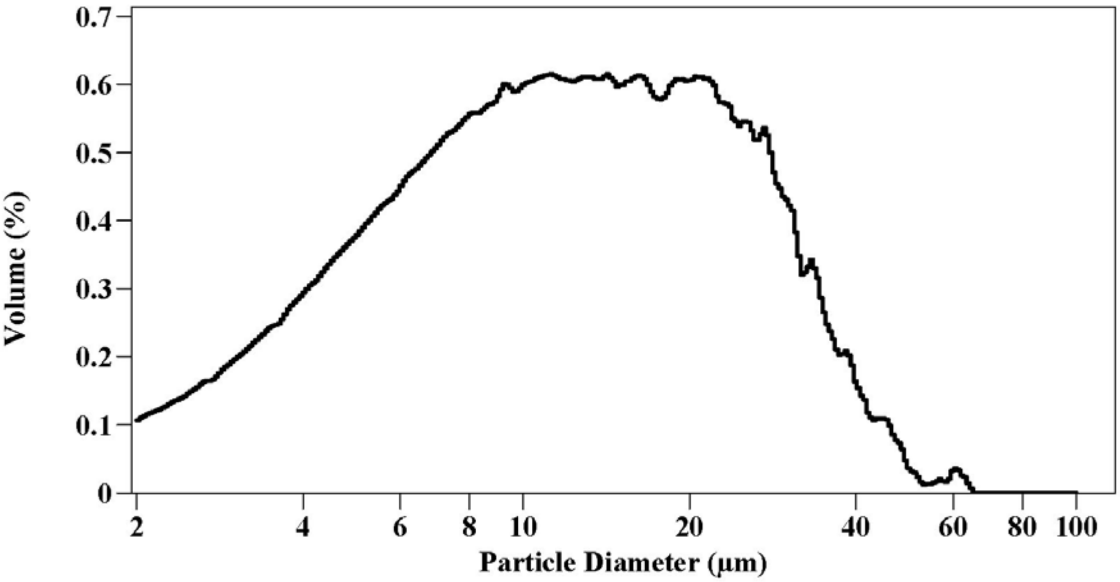

**Figure 119.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed stripper gin trash from the first stick machine.

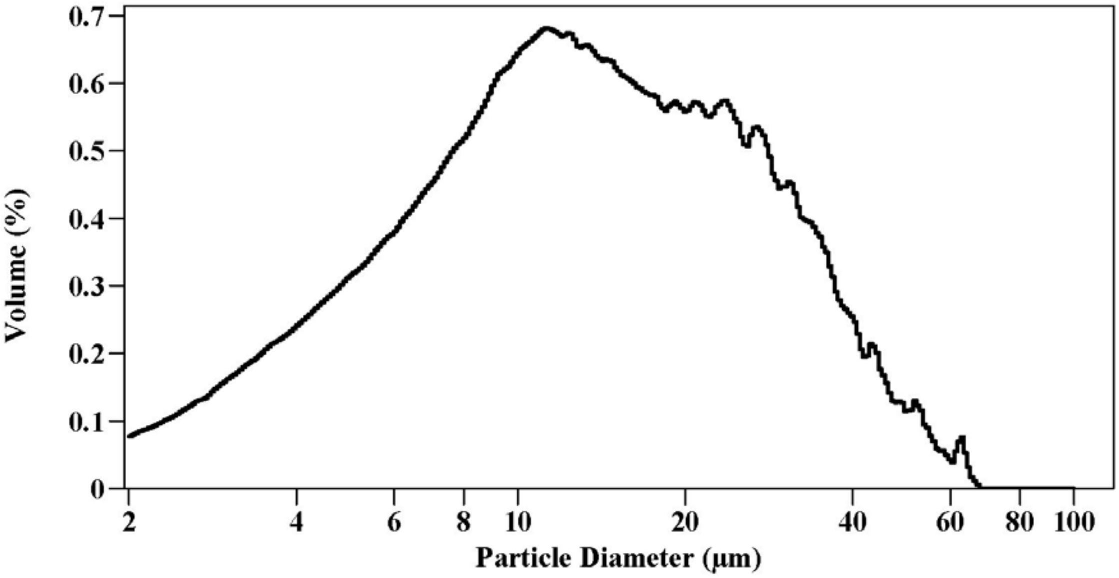

**Figure 120.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed picker gin trash from the second cylinder cleaner.

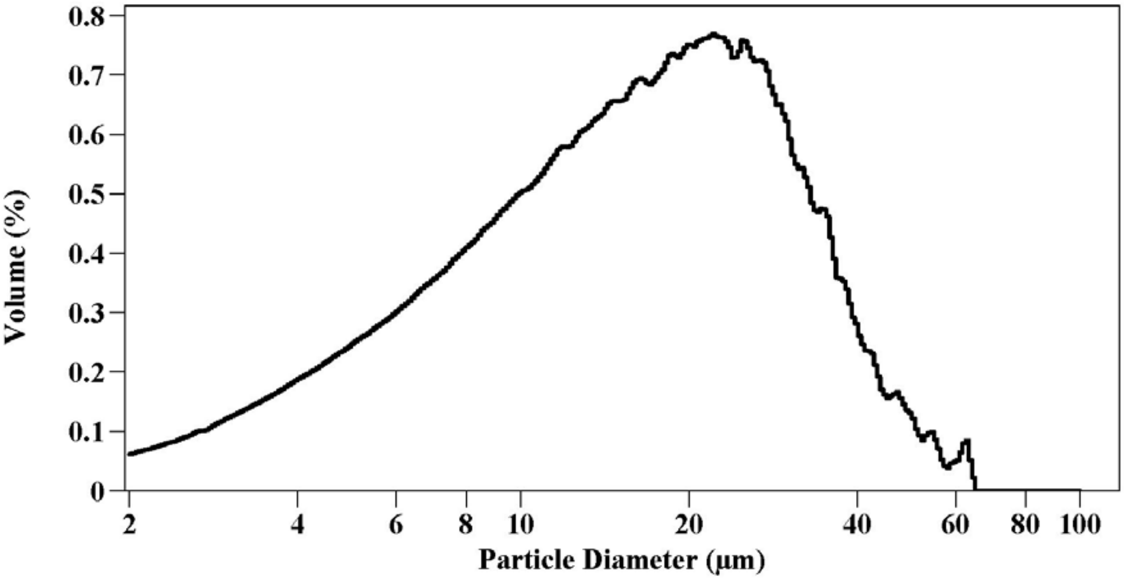

**Figure 121.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed stripper gin trash from the second cylinder cleaner.

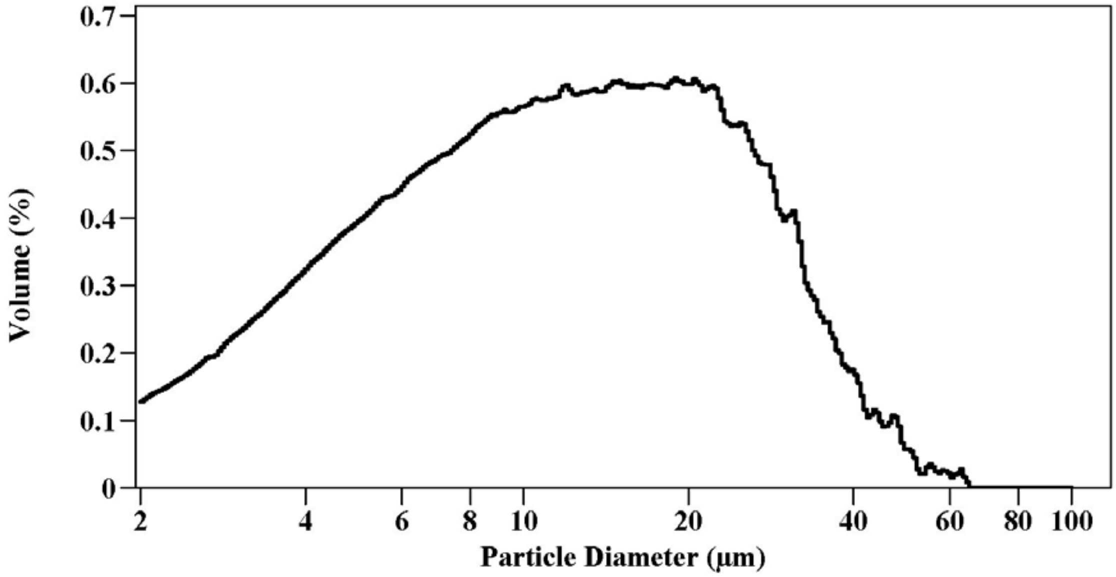

**Figure 122.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed stripper gin trash from the second stick machine.

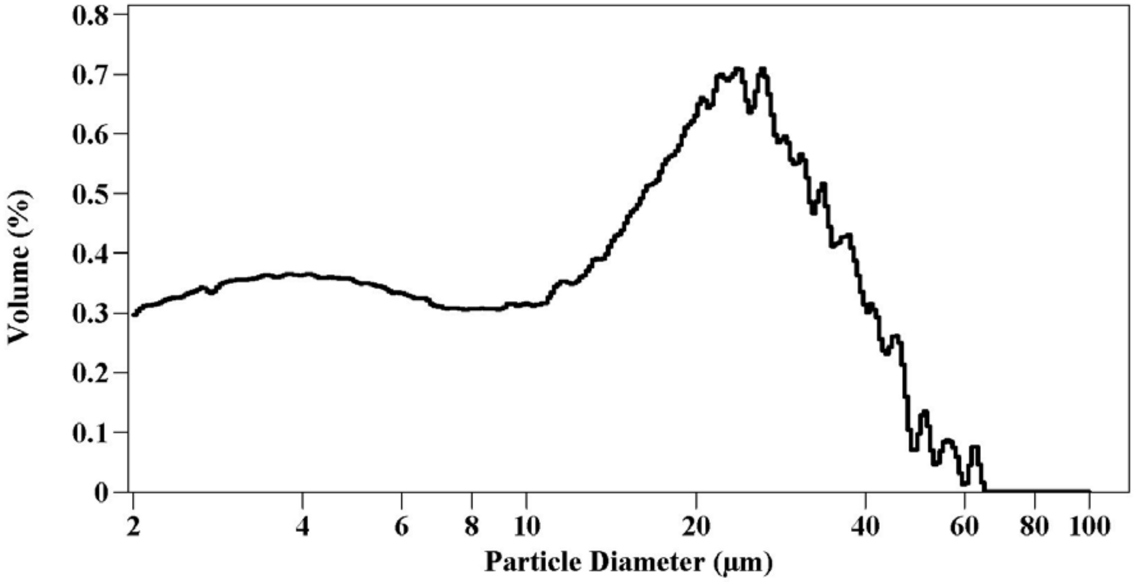

**Figure 123.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed picker gin trash from the gin stand and lint cleaners.

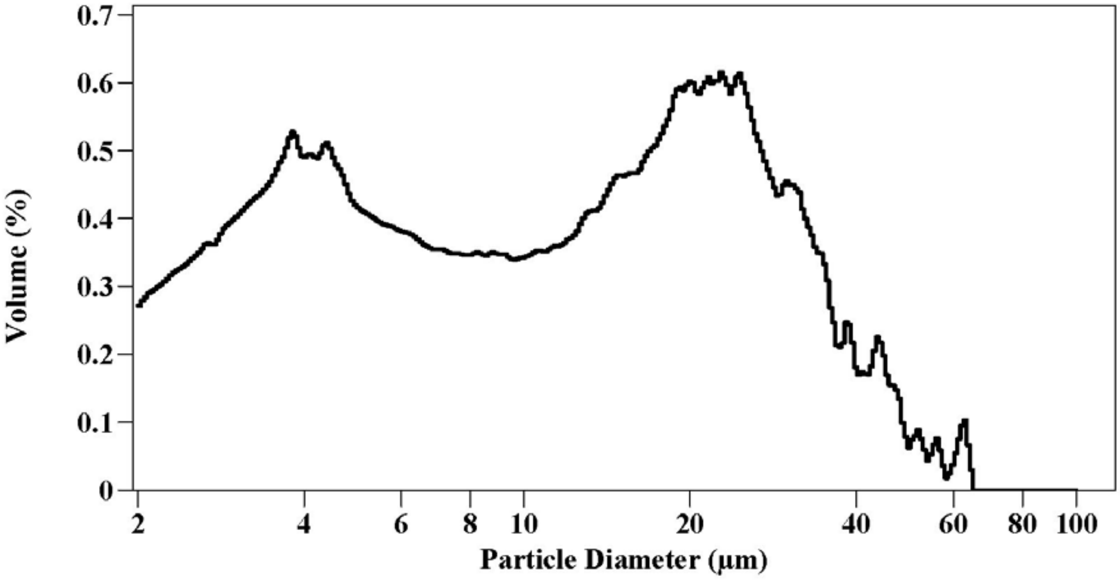

**Figure 124.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed stripper gin trash from the lint cleaners.

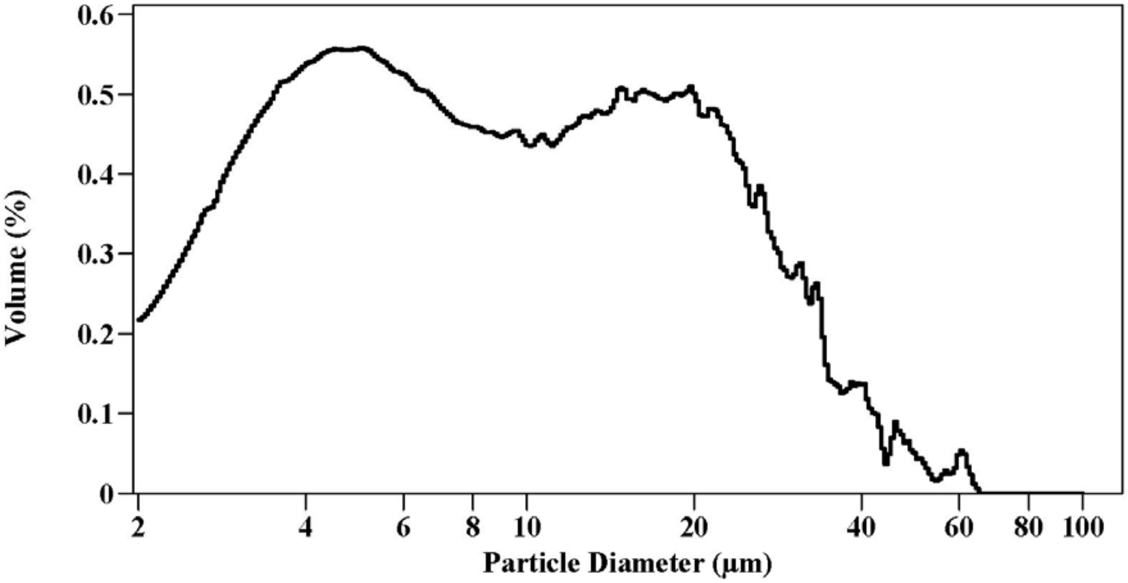

**Figure 125.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from electrolyte-washed stripper gin trash from the lint cleaners.

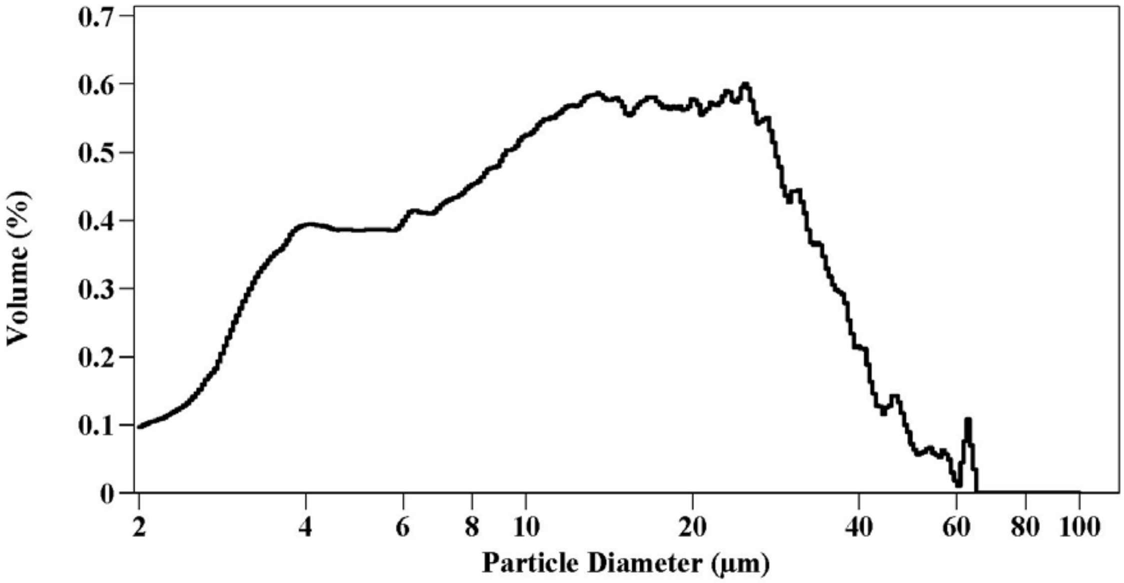

**Figure 126.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed picker and stripper gin trash, all systems combined (i.e. master cyclone).

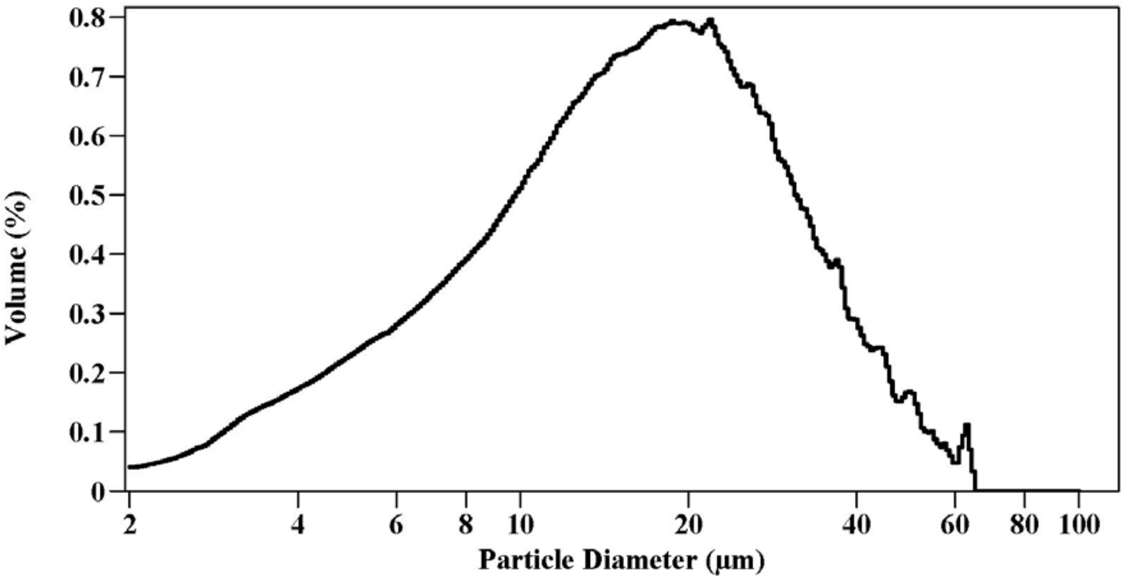

**Figure 127.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from electrolyte-washed picker and stripper gin trash, all systems combined (i.e. master cyclone).

# APPENDIX F

# PARTICLE SIZE DISTRIBUTION CHARACTERISTICS OF STRIPPER AND PICKER COTTON GIN TRASH, SEGREGATED BY SIMILAR PROCESS STREAMS, GINNED AT THE USDA-ARS COTTON PRODUCTION AND PROCESSING RESEARCH UNIT IN LUBBOCK, TX

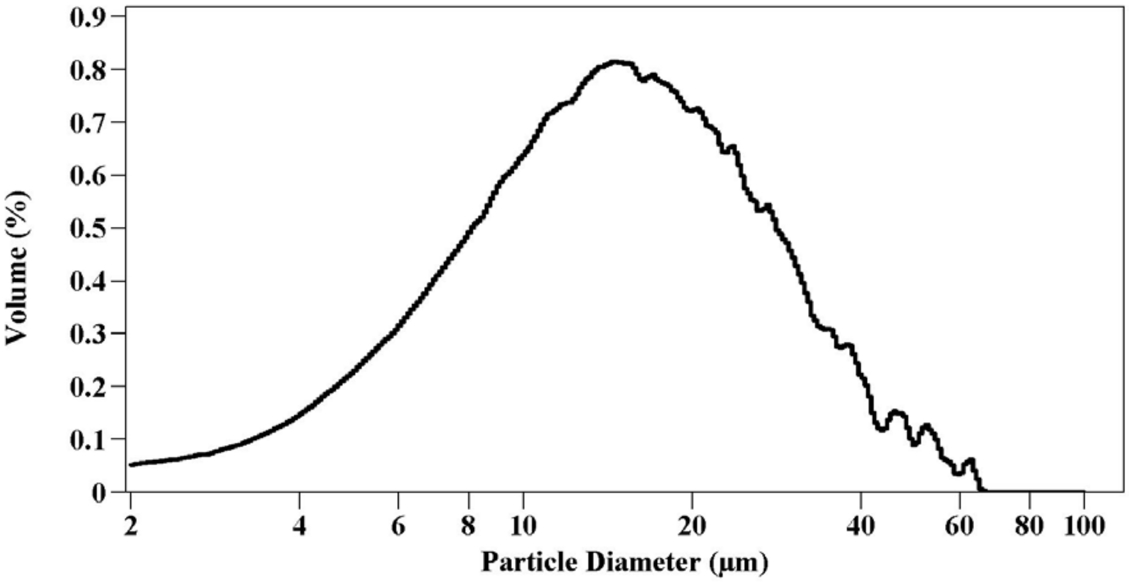

**Figure 128.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed, non-field cleaned stripper gin trash from the unloading system (Paymaster HS-26 cotton variety).

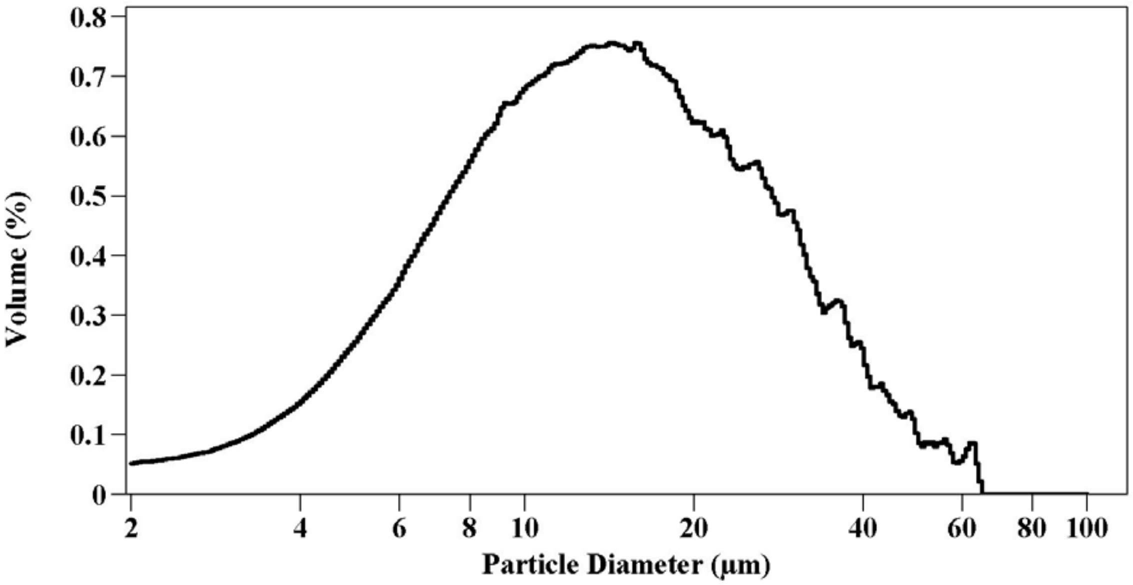

**Figure 129.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed, non-field cleaned stripper gin trash from the unloading system (Paymaster HS-200 cotton variety).

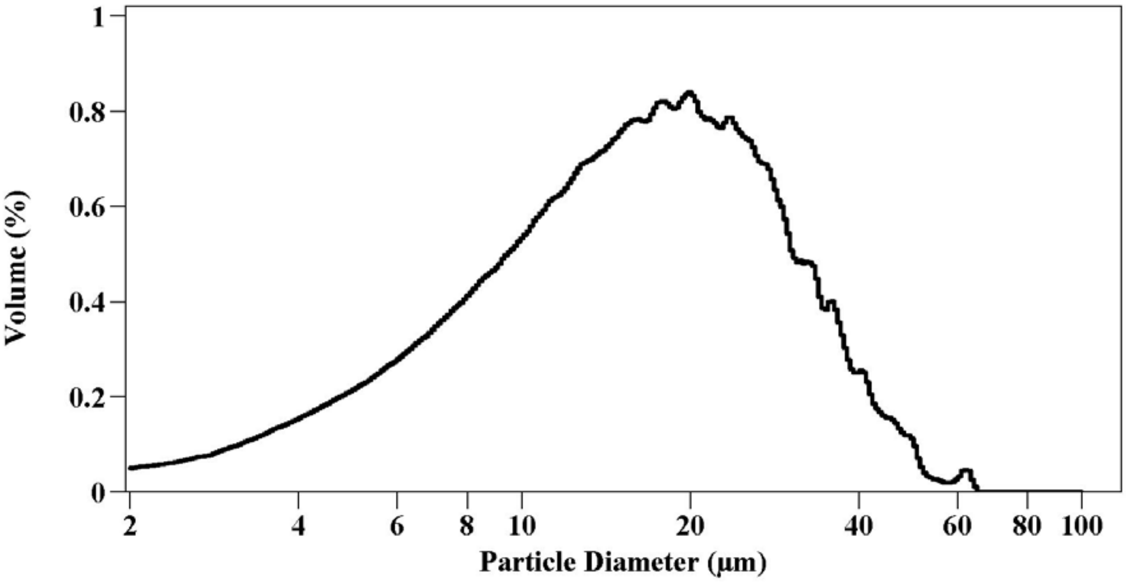

**Figure 130.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed field cleaned stripper gin trash from the feeder and gin stand (Paymaster HS-26 cotton variety).

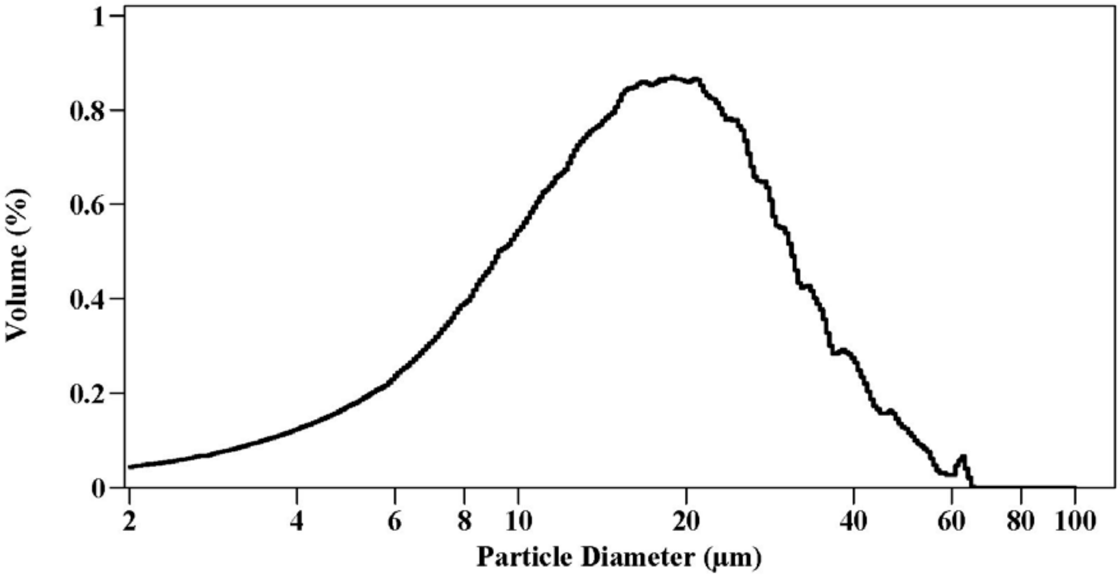

**Figure 131.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed, non-field cleaned stripper gin trash from the feeder and gin stand (Paymaster HS-26 cotton variety).

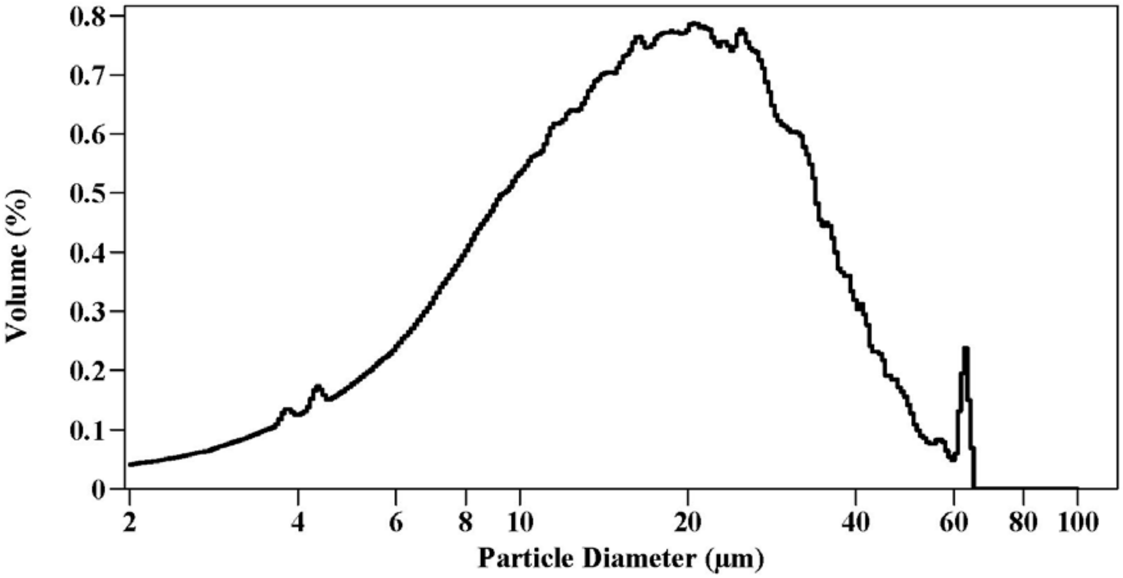

**Figure 132.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed, non-field cleaned stripper gin trash from the feeder and gin stand (Paymaster HS-200 cotton variety).

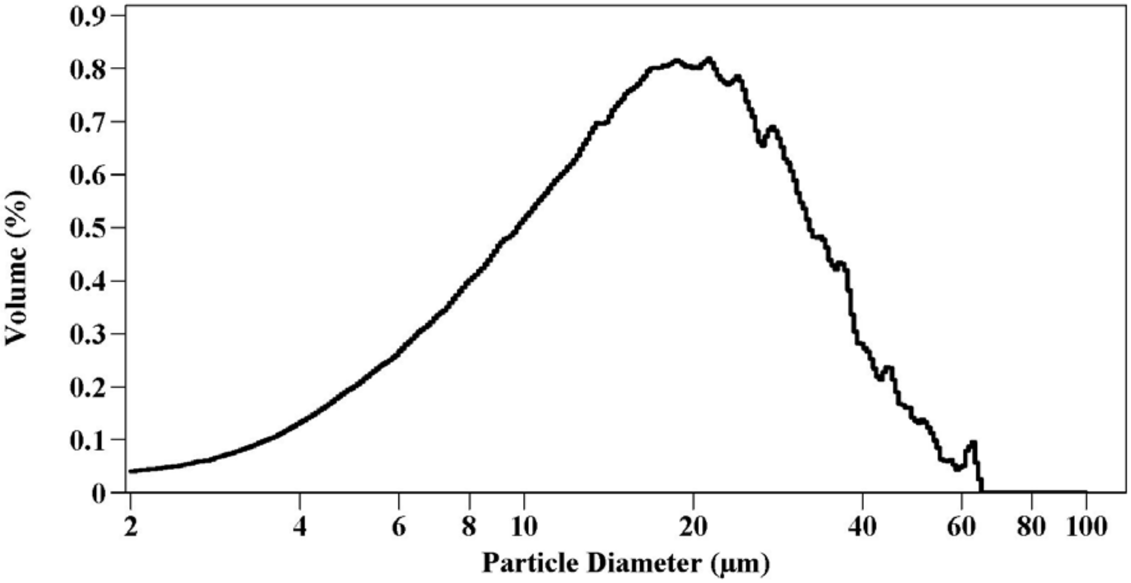

**Figure 133.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed, non-field cleaned stripper gin trash from the incline cleaners (Paymaster HS-26 cotton variety).

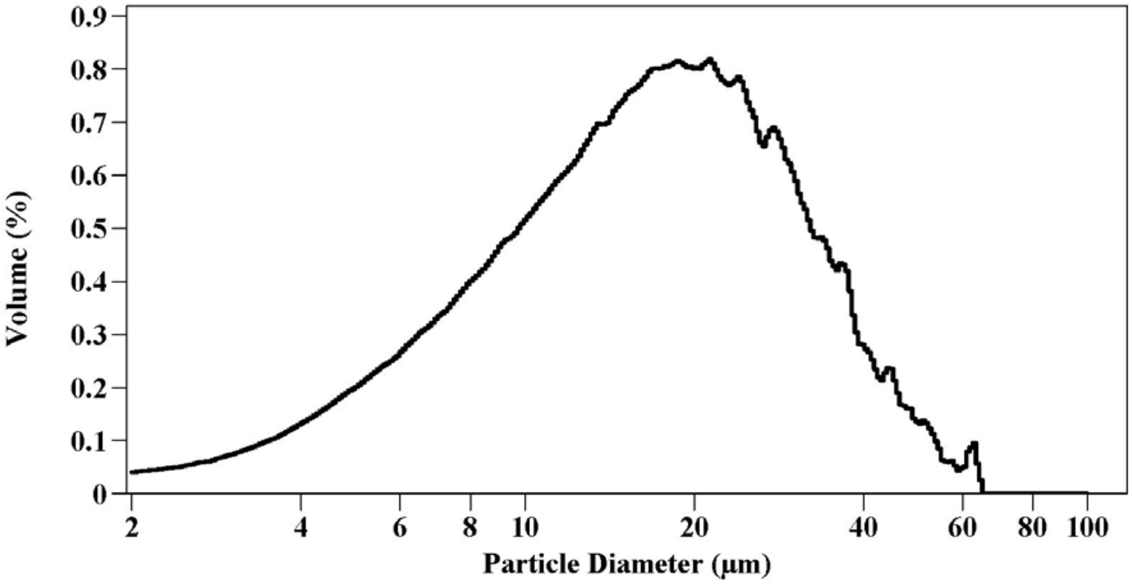

**Figure 134.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed field cleaned stripper gin trash from the incline cleaners (Paymaster HS-200 cotton variety).

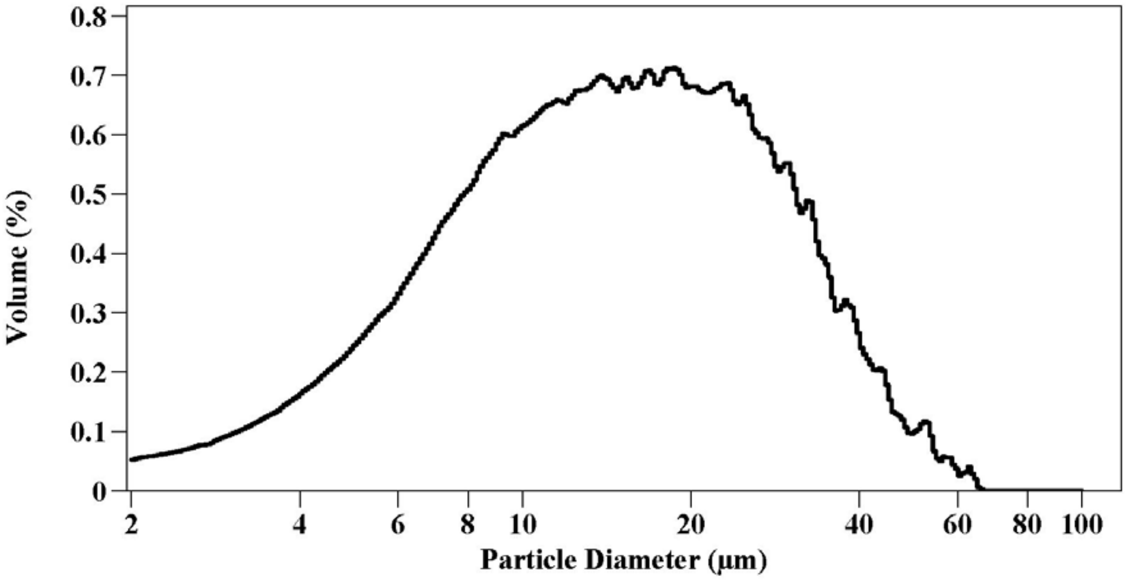

**Figure 135.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed, non-field cleaned stripper gin trash from the incline cleaners (Paymaster HS-200 cotton variety).

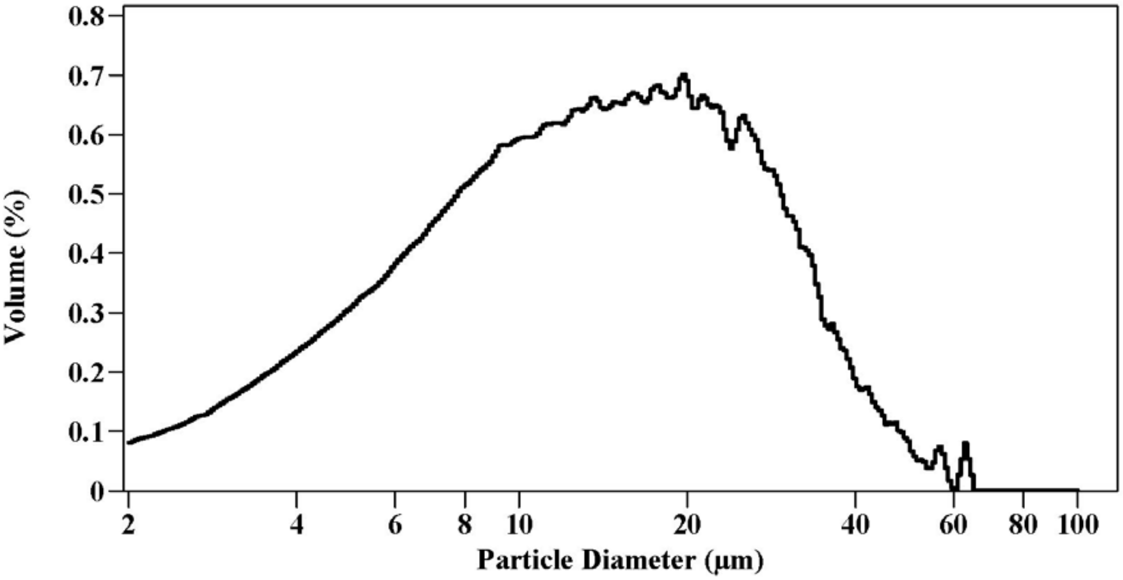

**Figure 136.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed field cleaned stripper gin trash from the extractors (Paymaster HS-200 cotton variety).

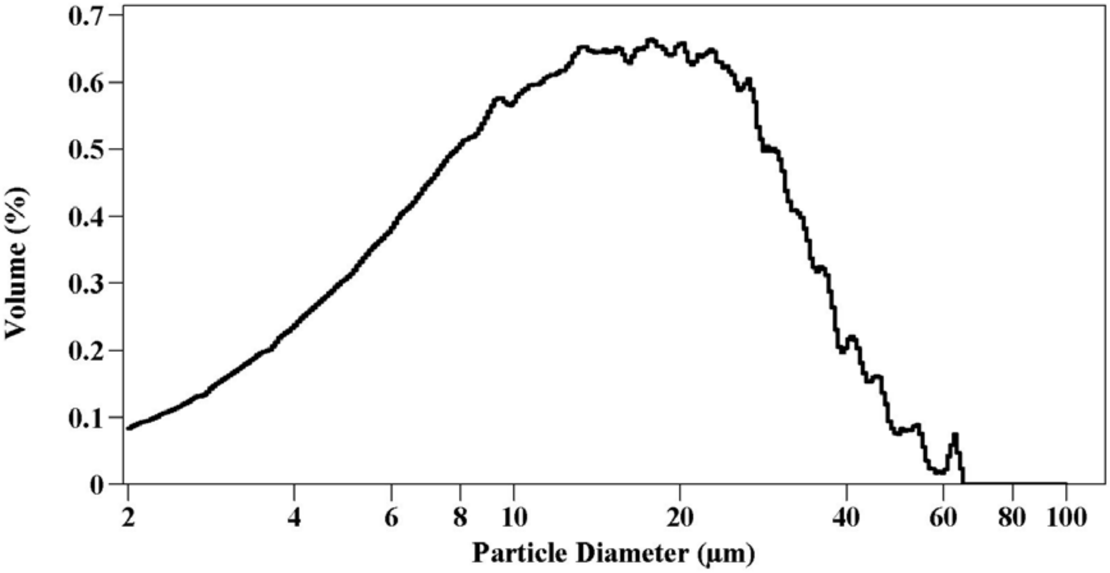

**Figure 137.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed, non-field cleaned stripper gin trash from the extractors (Paymaster HS-200 cotton variety).

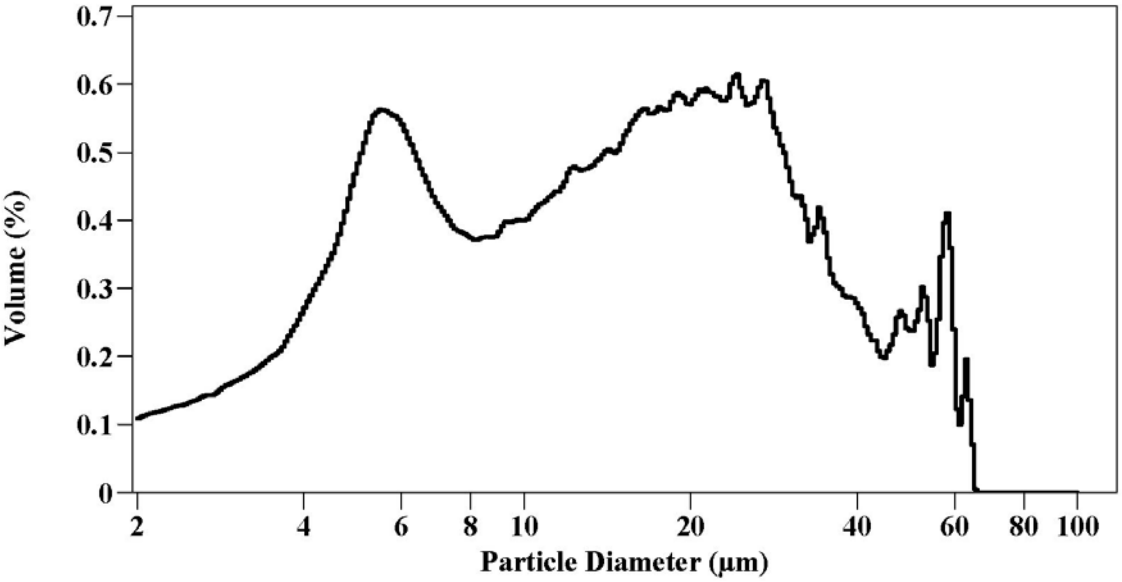

**Figure 138.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed field cleaned stripper gin trash from the lint cleaners (Paymaster HS-26 cotton variety).

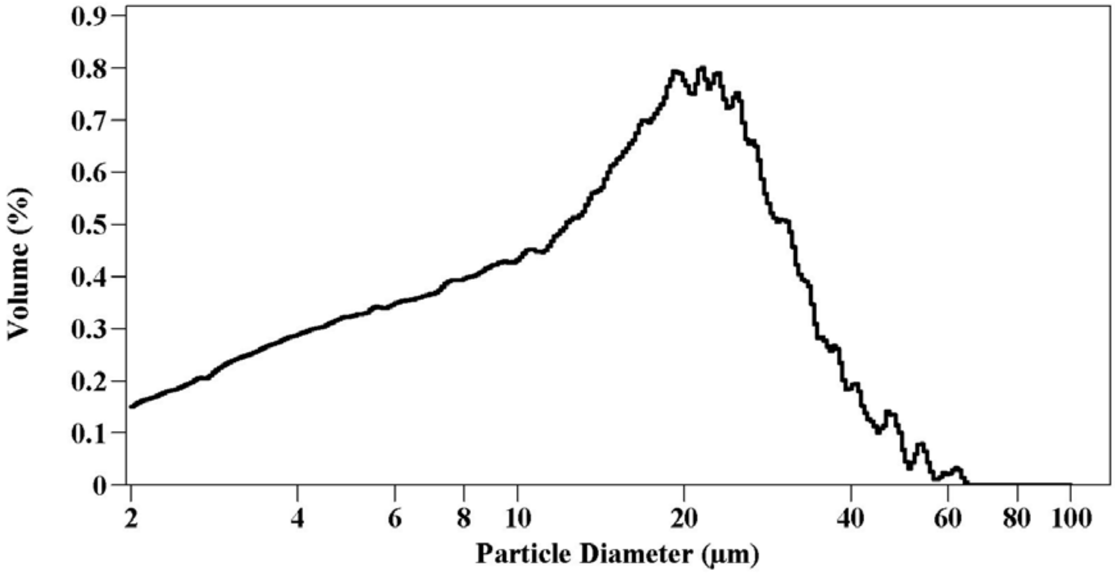

**Figure 139.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed, non-field cleaned stripper gin trash from the lint cleaners (Paymaster HS-200 cotton variety).

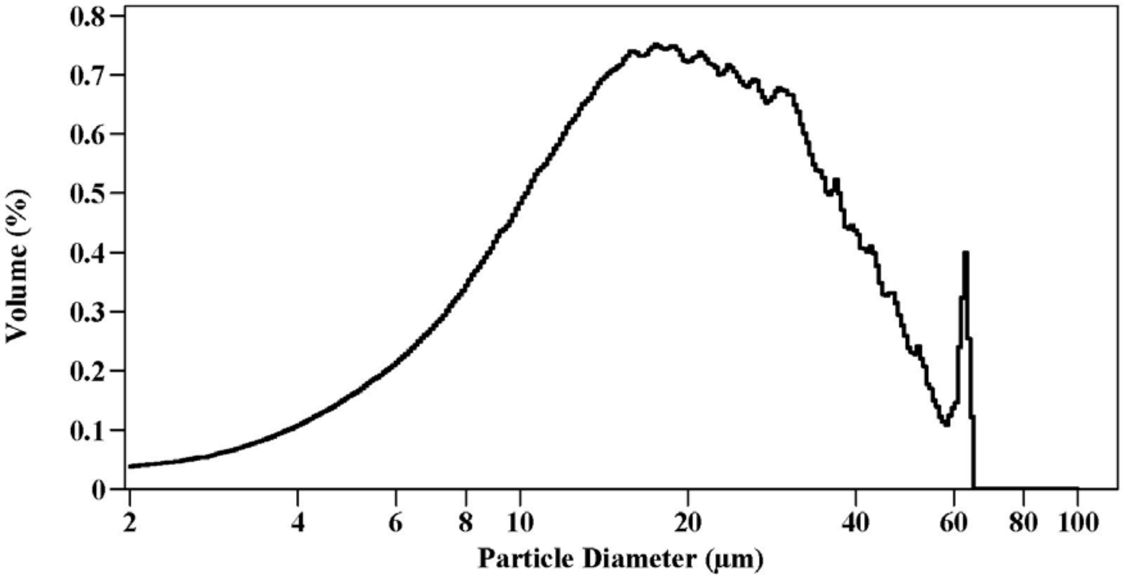

**Figure 140.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed field cleaned stripper gin trash, all systems combined (Paymaster HS-26 cotton variety).

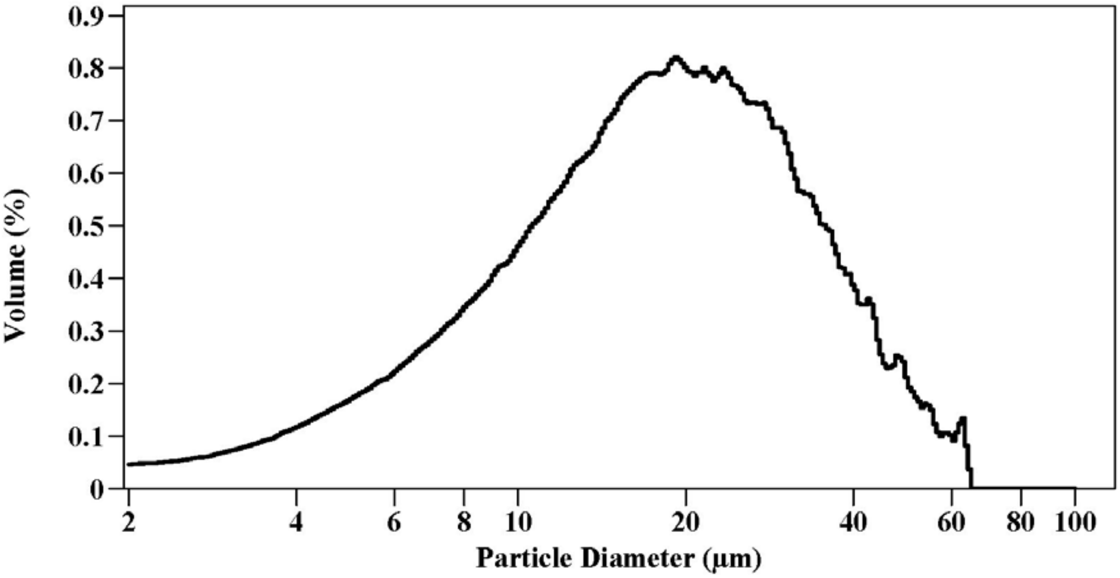

**Figure 141.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed, non-field cleaned stripper gin trash, all systems combined (Paymaster HS-26 cotton variety).

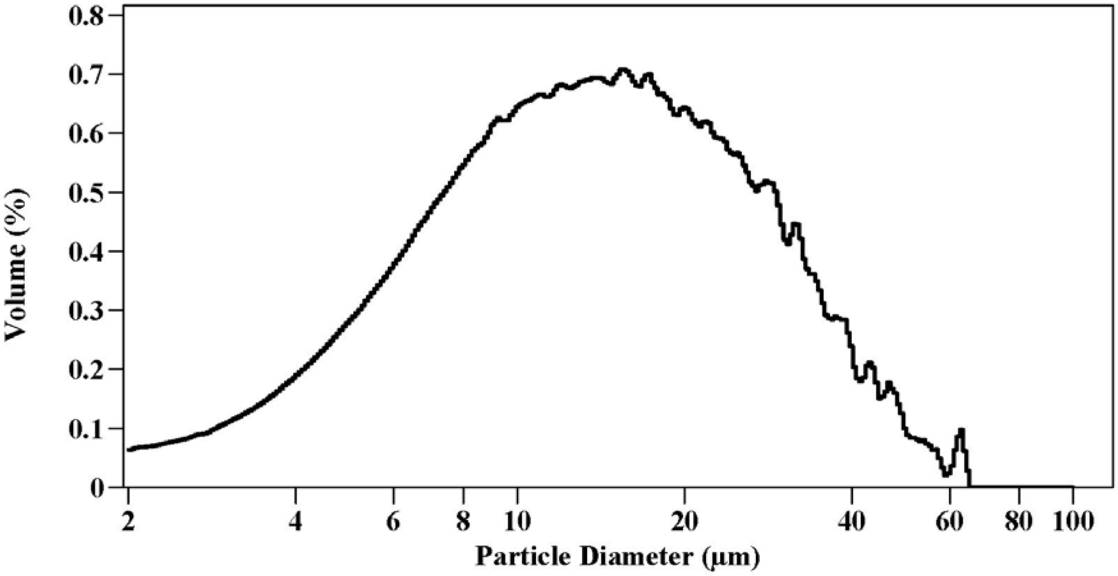

**Figure 142.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed field cleaned stripper gin trash, all systems combined (Paymaster HS-200 cotton variety).

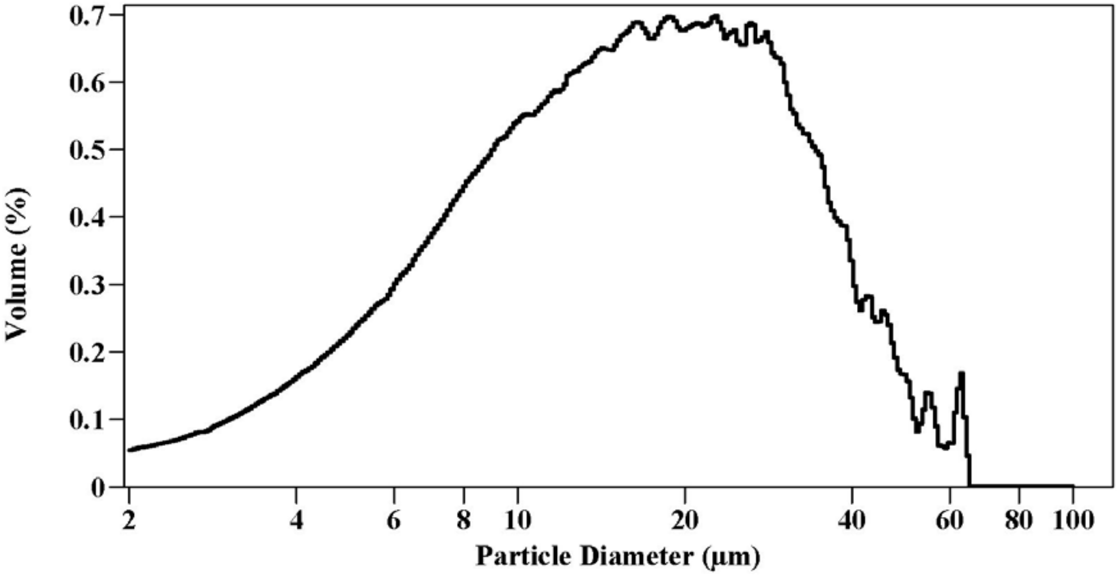

**Figure 143.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from air-washed, non-field cleaned stripper gin trash, all systems combined (Paymaster HS-200 cotton variety).

# APPENDIX G

# DATA ASSOCIATED WITH STACK SAMPLING CONDUCTED ON A CALIFORNIA COTTON GIN’S 1$^{ST}$ STAGE OF PRE-CLEANING FOR 1$^{ST}$ AND 2$^{ND}$ PICKED PIMA COTTON

**Table 35.** Idria Gin #1 source sampling parameter values determined by $Air_x$ testing and used in calculating emission factors.

| Harvest | Run No. | Sampling Time (min) | Nozzle Area ($cm^2$) | Isokinetic Variation (%) | Processing Rate (bales/h) |
|---|---|---|---|---|---|
| 1st Pick | | | | | |
| | 1 | 44.16 | 0.356 | 92.2 | 17.94 |
| | 2 | 45.18 | 0.359 | 96.5 | 14.76 |
| | 3 | 45.66 | 0.356 | 99.5 | 14.76 |
| | 4 | 44.40 | 0.359 | 94.8 | 17.75 |
| | 5 | 44.99 | 0.348 | 96.2 | 17.43 |
| | 6 | 48.80 | 0.359 | 98.3 | 16.96 |
| 2nd Pick | | | | | |
| | 1 | 46.28 | 0.348 | 93.4 | 19.25 |
| | 2 | 49.99 | 0.359 | 88.0 | 16.29 |
| | 3 | 40.52 | 0.275 | 117.7 | 17.04 |
| | 4 | 43.72 | 0.348 | 95.7 | 14.87 |
| | 5 | 51.40 | 0.368 | 115.8 | 14.23 |
| | 6 | 39.83 | 0.348 | 97.1 | 16.50 |

**Table 36.** Source sampling component gravimetric weights for Idria Gin #1 producing 1st and 2nd pick Pima cotton as determined by $Air_x$ testing and PM percentages determined by Coulter Counter analyses.

| | | Cyclone Wash | | | Post-Cyclone Wash | | | Filter | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Harvest | Run No. | Wt. (g) | $PM_{10}$ (%) | $PM_{2.5}$ (%) | Wt. (g) | $PM_{10}$ (%) | $PM_{2.5}$ (%) | Wt. (g) | $PM_{10}$ (%) | $PM_{2.5}$ (%) |
| 1st Pick | | | | | | | | | | |
| | 1 | 0.0387 | 5.39 | 0.0005 | 0.0083 | 17.39 | 0.10 | 0.0539 | 57.38 | 0.19 |
| | 2 | 0.0334 | 6.45 | 0.0006 | 0.0168 | 20.30 | 0.19 | 0.0462 | 66.00 | 0.39 |
| | 3 | 0.0377 | 4.03 | 0.0001 | 0.0076 | 15.44 | 0.05 | 0.0602 | 57.68 | 0.15 |
| | 4 | 0.0407 | 9.66 | 0.0033 | 0.0109 | 13.78 | 0.04 | 0.0558 | 68.59 | 0.45 |
| | 5 | 0.0465 | 8.35 | 0.0024 | 0.0086 | 31.37 | 0.59 | 0.0559 | 64.19 | 0.29 |
| | 6 | 0.0414 | 26.15 | 0.0738 | 0.0107 | 24.82 | 0.21 | 0.0730 | 81.85 | 0.78 |
| 2nd Pick | | | | | | | | | | |
| | 1 | 0.0288 | 21.62 | 0.0770 | 0.0061 | 24.13 | 0.12 | 0.0252 | 35.83 | 0.04 |
| | 2 | 0.0492 | 12.49 | 0.0115 | 0.0066 | 25.16 | 0.12 | 0.0531 | 64.11 | 0.33 |
| | 3 | 0.0384 | 13.47 | 0.0102 | 0.0090 | 19.00 | 0.03 | 0.0344 | 67.57 | 0.43 |
| | 4 | 0.0327 | 13.75 | 0.0202 | 0.0106 | 18.76 | 0.09 | 0.0351 | 55.91 | 0.13 |
| | 5 | 0.0408 | 10.90 | 0.0061 | 0.0110 | 26.55 | 0.10 | 0.0390 | 72.45 | 0.62 |
| | 6 | 0.0296 | 10.31 | 0.0240 | 0.0123 | 21.32 | 0.07 | 0.0386 | 67.69 | 0.32 |

**Table 37.** Idria Gin #1's 1st seed cotton cleaning and drying system's exhaust emission factors as determined by source sampling and Coulter Counter analysis.

| | | Source Sampling | | | Coulter Counter Analysis (True) | | | |
|---|---|---|---|---|---|---|---|---|
| Harvest | Run No. | TSP** | $PM_{10}$** | $PM_{10}$/TSP Ratio | $PM_{10}$** | $PM_{2.5}$** | $PM_{10}$/TSP Ratio | $PM_{2.5}$/TSP Ratio |
| 1st Pick | | | | | | | | |
| | 1 | 0.0798 | 0.0492 | 61.7 % | 0.0272 | $0.9\ e^{-4}$ | 34.2 % | 0.11 % |
| | | (0.1760) | (0.1085) | | (0.0600) | $(1.9\ e^{-4})$ | | |
| | 2 | 0.0860 | 0.0562 | 65.3 % | 0.0321 | $1.9\ e^{-4}$ | 37.4 % | 0.22 % |
| | | (0.1897) | (0.1239) | | (0.0707) | $(4.2\ e^{-4})$ | | |
| | 3 | 0.0908 | 0.0584 | 64.3 % | 0.0322 | $0.8\ e^{-4}$ | 35.5% | 0.09 % |
| | | (0.2001) | (0.1287) | | (0.0710) | $(1.8\ e^{-4})$ | | |
| | 4 | 0.0822 | 0.0511 | 62.2 % | 0.0335 | $2.0\ e^{-4}$ | 40.7 % | 0.24 % |
| | | (0.1813) | (0.1127) | | (0.0738) | $(4.4\ e^{-4})$ | | |
| | 5 | 0.0870 | 0.0505 | 58.1 % | 0.0332 | $1.7\ e^{-4}$ | 38.3 % | 0.19 % |
| | | (0.1918) | (0.1114) | | (0.0732) | $(3.7\ e^{-4})$ | | |
| | 6 | 0.0881 | 0.0589 | 66.8 % | 0.0515 | $4.4\ e^{-4}$ | 58.5 % | 0.50 % |
| | | (0.1942) | (0.1298) | | (0.1136) | $(9.6\ e^{-4})$ | | |
| | Average | 0.0857 | 0.0541 | 63.1 % | 0.0350 | $1.9\ e^{-4}$ | 40.8 % | 0.22 % |
| | | (0.1889) | (0.1192) | | (0.0771) | $(4.3\ e^{-4})$ | | |
| 2nd Pick | | | | | | | | |
| | 1 | 0.0426 | 0.0222 | 52.1 % | 0.0119 | $0.3\ e^{-4}$ | 27.8 % | 0.06 % |
| | | (0.0940) | (0.0490) | | (0.0262) | $(0.6\ e^{-4})$ | | |
| | 2 | 0.0872 | 0.0478 | 54.8 % | 0.0334 | $1.5\ e^{-4}$ | 38.4 % | 0.17 % |
| | | (0.1922) | (0.1054) | | (0.0737) | $(3.3\ e^{-4})$ | | |
| | 3 | 0.0752 | 0.0399 | 53.1 % | 0.0277 | $1.4\ e^{-4}$ | 36.8 % | 0.19 % |
| | | (0.1657) | (0.0879) | | (0.0610) | $(3.1\ e^{-4})$ | | |
| | 4 | 0.0744 | 0.0434 | 58.3 % | 0.0248 | $0.6\ e^{-4}$ | 33.3 % | 0.08 % |
| | | (0.1641) | (0.0956) | | (0.0546) | $(1.3\ e^{-4})$ | | |
| | 5 | 0.0801 | 0.0441 | 55.1 % | 0.0314 | $2.2\ e^{-4}$ | 39.2 % | 0.28 % |
| | | (0.1765) | (0.0972) | | (0.0692) | $(4.9\ e^{-4})$ | | |
| | 6 | 0.0746 | 0.0471 | 63.2 % | 0.0294 | $1.3\ e^{-4}$ | 39.5 % | 0.17 % |
| | | (0.1644) | (0.1039) | | (0.0648) | $(2.8\ e^{-4})$ | | |
| | Average | 0.0723 | 0.0407 | 56.1 % | 0.0264 | $1.2\ e^{-4}$ | 35.9 % | 0.16 % |
| | | (0.1595) | (0.0898) | | (0.0582) | $(2.7\ e^{-4})$ | | |
| MSE | | 0.0006 | 0.0003 | 0.1313 | 0.0003 | $5.3\ e^{-8}$ | 0.5083 | $1.4\ e^{-}6$ |
| p-value*** | | 0.0668 | 0.0108 | 0.0076 | 0.0977 | 0.2645 | 0.2615 | 0.3621 |
| F-value | | 4.23 | 9.77 | 11.10 | 3.34 | 1.40 | 1.42 | 0.91 |

Emission factors correspond to only the one cyclone tested; these emission factors must be multiplied by 4 to determine the total emission factor for the #1 pre-cleaning system.

** TSP, $PM_{10}$, and $PM_{2.5}$ emission factors reported in kg of PM emitted per bale of lint cotton produced (lb of PM emitted per bale of lint cotton produced).

*** p-value is the test of significance of means being equal in the PROC Mixed procedure of SAS. Any p-value greater than 0.05 was considered to be non-significant.

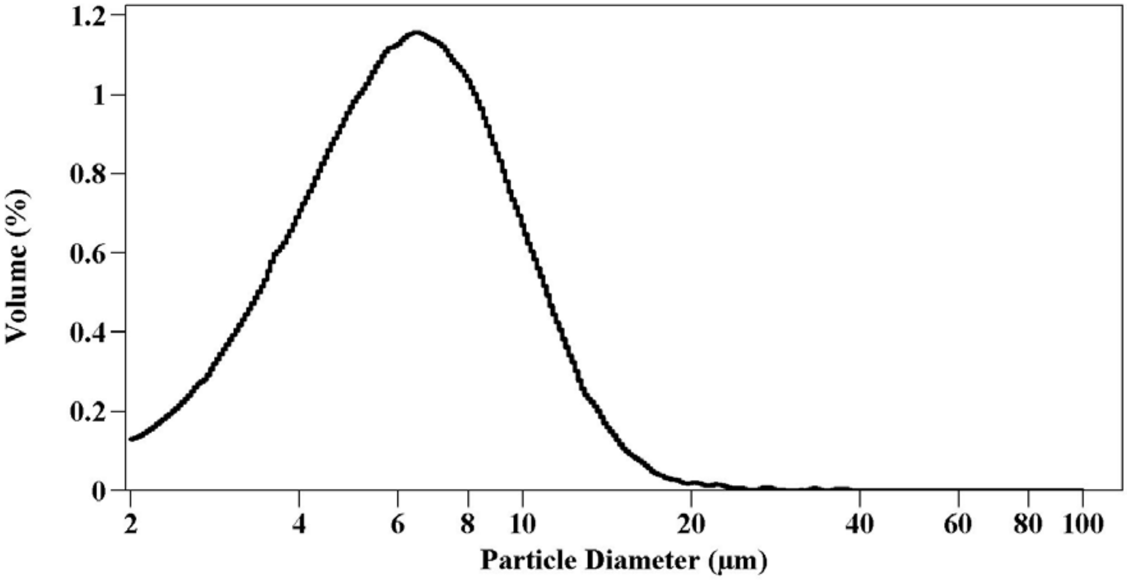

**Figure 144.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 201a glass fiber filter obtained from stack sampling conducted on a #1 pre-cleaning system exhaust while processing 1st pick Pima cotton.

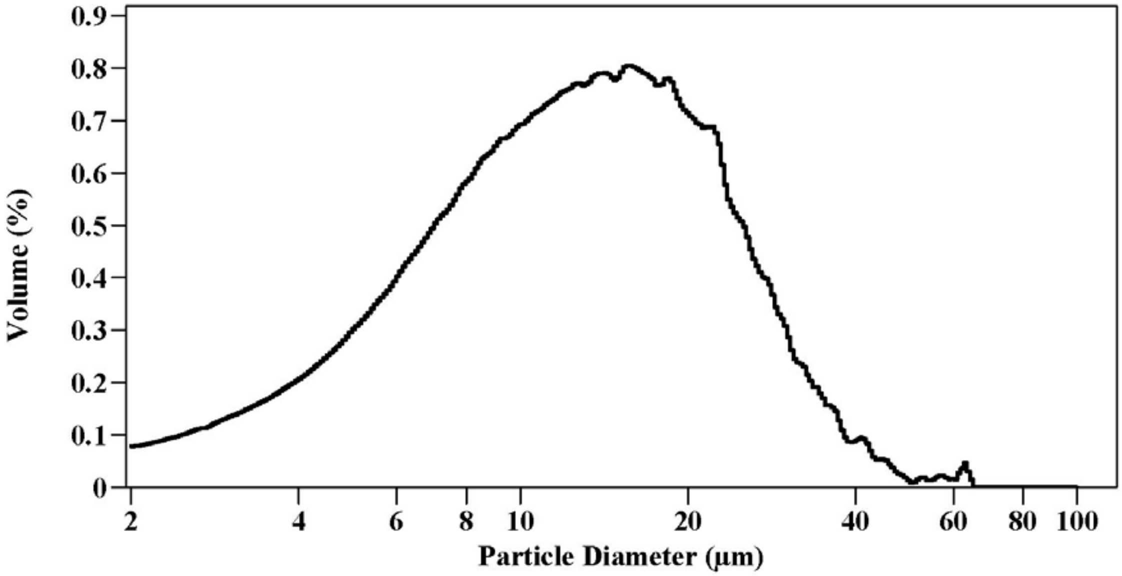

**Figure 145.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the post-cyclone wash obtained from Method 201a stack sampling conducted on a #1 pre-cleaning system exhaust while processing 1st pick Pima cotton.

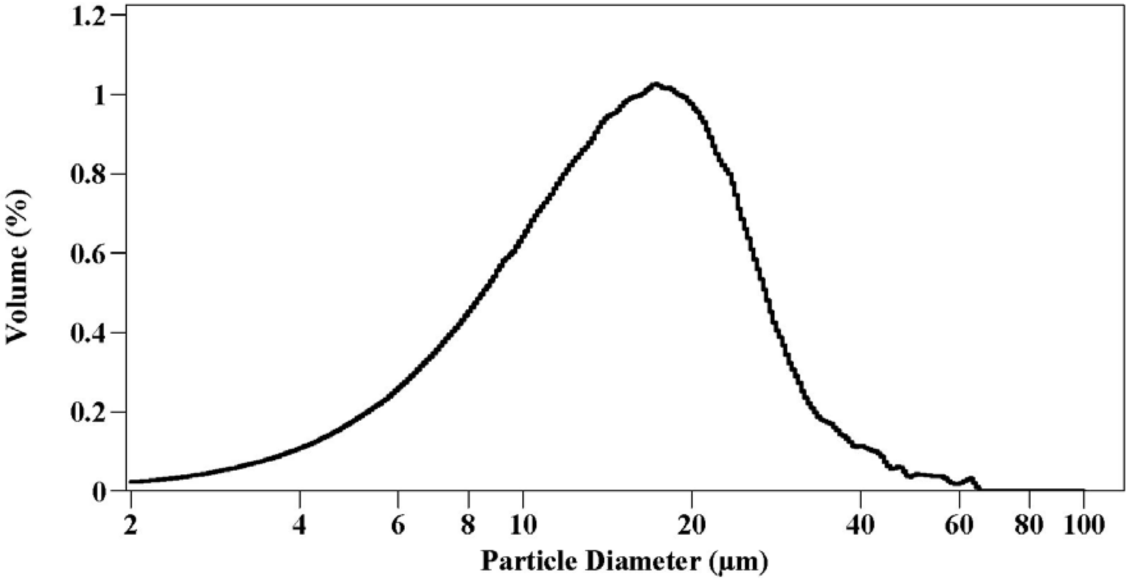

**Figure 146.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 201a stack sampling conducted on a #1 pre-cleaning system exhaust while processing 1st pick Pima cotton.

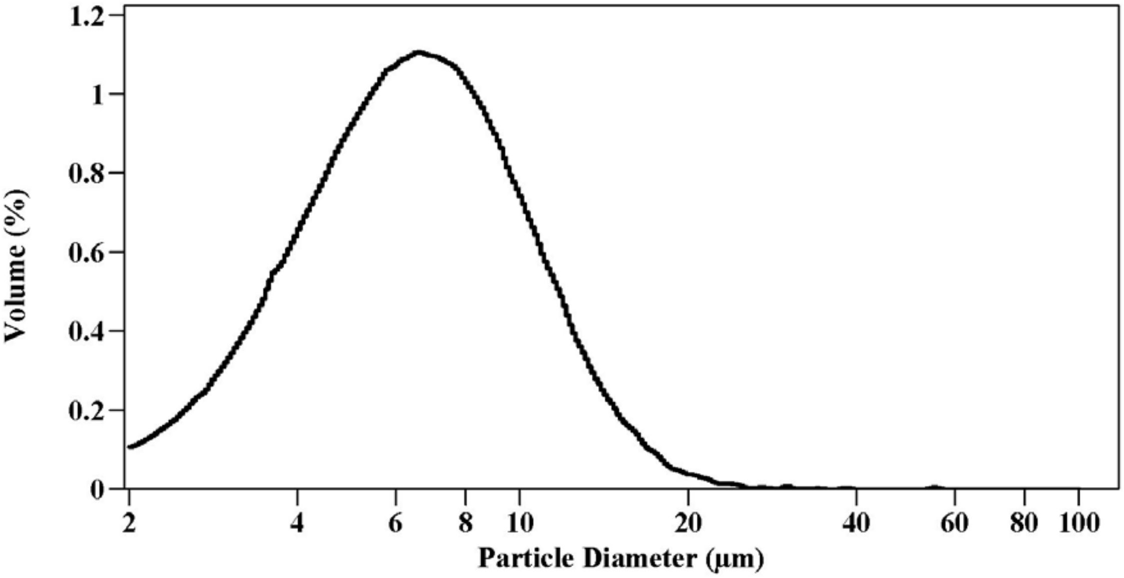

**Figure 147.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 201a glass fiber filter obtained from stack sampling conducted on a #1 pre-cleaning system exhaust while processing 2nd pick Pima cotton.

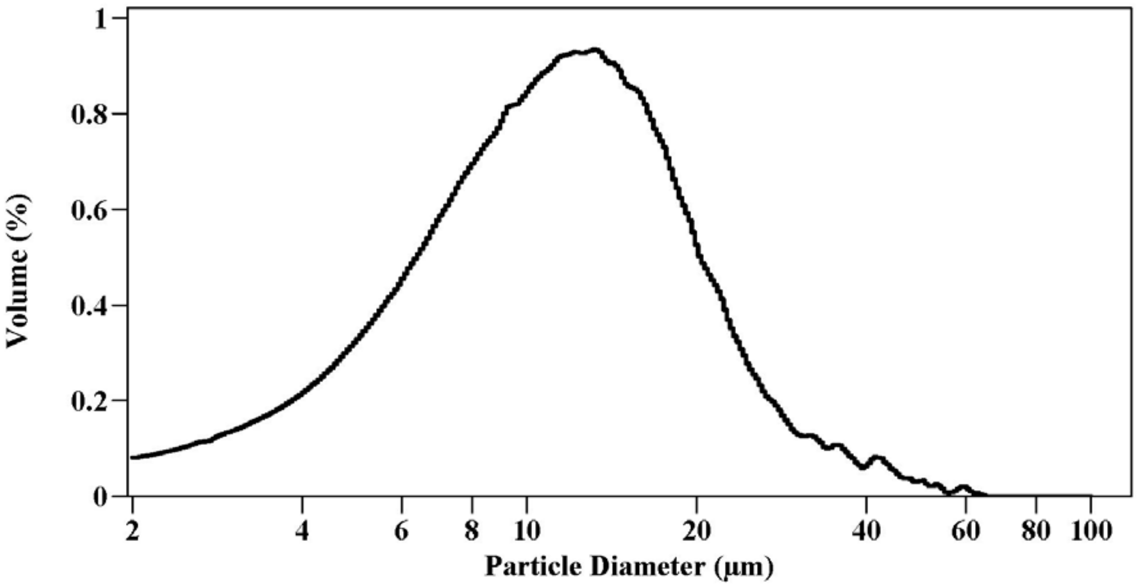

**Figure 148.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the post-cyclone wash obtained from Method 201a stack sampling conducted on a #1 pre-cleaning system exhaust while processing 2nd pick Pima cotton.

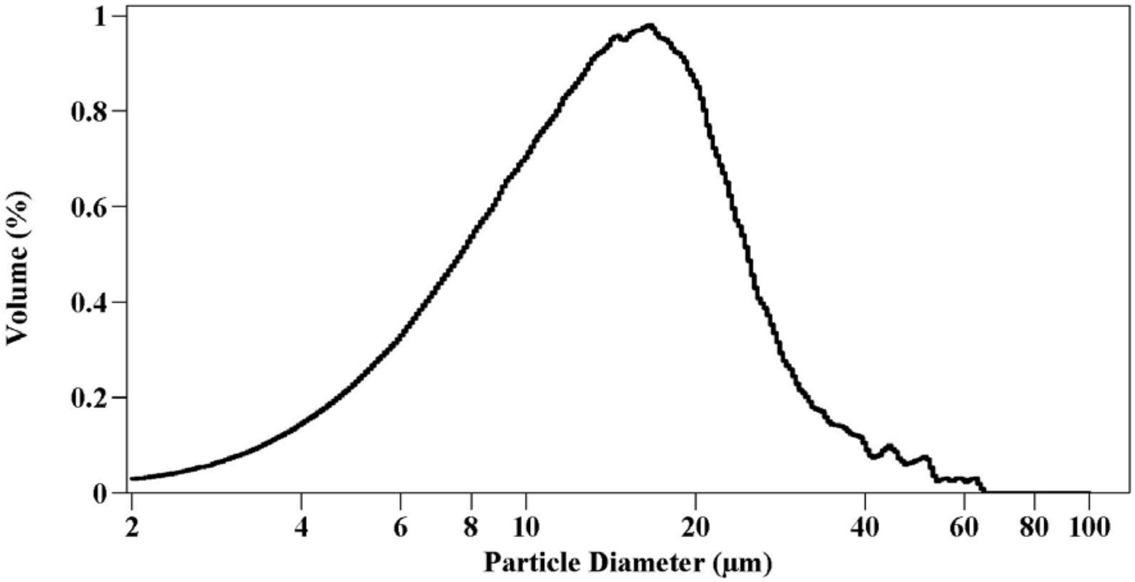

**Figure 149.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 201A stack sampling conducted on a #1 pre-cleaning system exhaust while processing 2nd pick Pima cotton.

# APPENDIX H

# DATA ASSOCIATED WITH STACK SAMPLING CONDUCTED ON VARIOUS PROCESS STREAM EXHAUSTS OF A NEW MEXICO COTTON GIN

**Table 38.** Mesa Gin source sampling parameter values determined by EEMC testing and used in calculating TSP emission factors.

| System | Run No. | Stack Area ($m^2$) | Nozzle Area ($cm^2$) | Isokinetic Variation (%) | Processing Rate (bales/h) |
|---|---|---|---|---|---|
| Unloading | 1 | 0.369 | 0.245 | 101.9 | 15.1 |
| | 2 | 0.348 | 0.245 | 101.1 | 15.1 |
| | 3 | 0.348 | 0.245 | 101.5 | 15.1 |
| #1 Pull | 1 | 0.292 | 0.245 | 101.6 | 14.9 |
| | 2 | 0.292 | 0.245 | 100.7 | 14.9 |
| | 3 | 0.292 | 0.245 | 101.4 | 14.9 |
| #2 Pull | 1 | 0.292 | 0.317 | 101.4 | 15.0 |
| | 2 | 0.292 | 0.317 | 101.2 | 15.0 |
| | 3 | 0.292 | 0.317 | 101.2 | 15.0 |
| #3A Pull | 1 | 0.130 | 0.317 | 101.7 | 16.0 |
| | 2 | 0.130 | 0.317 | 101.4 | 16.0 |
| | 3 | 0.130 | 0.317 | 101.5 | 16.0 |
| #3B Pull | 1 | 0.130 | 0.317 | 101.5 | 17.2 |
| | 2 | 0.130 | 0.317 | 101.4 | 17.2 |
| | 3 | 0.130 | 0.317 | 101.3 | 17.2 |
| Lint Basket Pull | 1 | 0.183 | 0.174 | 101.5 | 16.4 |
| | 2 | 0.183 | 0.174 | 101.4 | 16.4 |
| | 3 | 0.183 | 0.174 | 101.6 | 16.4 |

**Table 39.** Mesa Gin source sampling parameter values determined by EEMC testing and used in calculating $PM_{10}$ emission factors.

| System | Run No. | Stack Area ($m^2$) | Nozzle Area ($cm^2$) | Isokinetic* Variation (%) | Processing Rate (bales/h) |
|---|---|---|---|---|---|
| Unloading | 1 | 0.369 | 0.150 | 109.1 | 15.1 |
| | 2 | 0.369 | 0.150 | 119.6 | 15.1 |
| | 3 | 0.369 | 0.150 | 118.8 | 15.1 |
| #1 Pull | 1 | 0.292 | 0.199 | 87.2 | 14.9 |
| | 2 | 0.292 | 0.199 | 86.5 | 14.9 |
| | 3 | 0.292 | 0.199 | 85.6 | 14.9 |
| #2 Pull | 1 | 0.292 | 0.199 | 109.6 | 15.0 |
| | 2 | 0.292 | 0.199 | 109.4 | 15.0 |
| | 3 | 0.292 | 0.199 | 107.5 | 15.0 |
| #3A Pull | 1 | 0.130 | 0.150 | 129.7 | 16.0 |
| | 2 | 0.130 | 0.150 | 130.2 | 16.0 |
| | 3 | 0.130 | 0.155 | 124.4 | 16.0 |
| #3B Pull | 1 | 0.130 | 0.130 | 119.3 | 17.2 |
| | 2 | 0.130 | 0.130 | 114.2 | 17.2 |
| | 3 | 0.130 | 0.130 | 114.5 | 17.2 |
| Lint Basket Pull | 1 | 0.183 | 0.100 | 122.9 | 16.4 |
| | 2 | 0.183 | 0.100 | 121.9 | 16.4 |
| | 3 | 0.183 | 0.100 | 121.9 | 16.4 |

[*] Values calculated from EEMC reported data.

**Table 40.** Average particle size distribution characteristics determined by Coulter Counter analyses for the Mesa Gin exhausts sampled by EEMC in December of 2001.

| PSD Characteristic | Location | Method 201a Sampling | | | Method 5 Sampling | |
|---|---|---|---|---|---|---|
| | | Filter | Post-Cyclone | Cyclone | Filter | Wash |
| MMD (μm)* | | | | | | |
| | Unloading | 6.6 c | 8.9 b | 10.8 a | 9.4 b | 8.4 b |
| | # 1 Pull | 6.5 c | 15.0 a | 21.2 b | 7.1 d | 17.4 a |
| | # 2 Pull | 7.0 b | 8.2 b | 13.8 a | 7.8 c | 12.7 ab |
| | #3A Pull | 7.3 b | 10.9 b | 12.7 a | 7.4 cd | 8.7 b |
| | #3B Pull | 8.2 a | 10.5 b | 14.2 a | 10.8 a | 10.0 b |
| | Lint Basket Pull | 8.2 a | 11.2 ab | 14.9 a | 10.8 a | 16.2 a |
| | MSE | 0.083 | 7.948 | 14.633 | 0.164 | 12.900 |
| | p-value | < 0.0001 | 0.0001 | < 0.0001 | < 0.0001 | < 0.0001 |
| | F-value | 61.70 | 6.50 | 7.54 | 156.84 | 10.44 |
| GSD | | | | | | |
| | Unloading | 1.6 b | 1.8 ab | 1.8 c | 2.0 a | 1.8 b |
| | # 1 Pull | 1.5 b | 1.6 b | 2.1 a | 1.7 c | 2.1 a |
| | # 2 Pull | 1.6 b | 1.6 b | 1.9 b | 1.8 b | 1.7 b |
| | #3A Pull | 1.6 b | 2.0 a | 1.9 b | 1.8 b | 1.8 b |
| | #3B Pull | 1.8 a | 1.8 ab | 1.9 b | 2.1 a | 1.8 b |
| | Lint Basket Pull | 1.8 a | 2.0 a | 1.9 bc | 2.1 a | 1.8 b |
| | MSE | 0.003 | 0.017 | 0.009 | 0.009 | 0.022 |
| | p-value | < 0.0001 | < 0.0001 | < 0.0001 | < 0.0001 | 0.0004 |
| | F-value | 40.00 | 10.35 | 10.36 | 34.76 | 5.61 |
| PM$_{10}$ (%) | | | | | | |
| | Unloading | 83.4 ab | 61.9 a | 45.5 a | 53.4 d | 63.4 a |
| | # 1 Pull | 85.2 a | 31.3 b | 15.1 b | 75.1 a | 25.8 bc |
| | # 2 Pull | 79.0 bc | 65.6 a | 34.9 ab | 65.7 c | 44.5 ab |
| | #3A Pull | 74.9 c | 46.8 ab | 37.0 ab | 70.2 b | 60.0 a |
| | #3B Pull | 63.3 d | 48.5 ab | 32.7 ab | 46.0 e | 50.4 a |
| | Lint Basket Pull | 64.4 d | 45.1 ab | 32.8 ab | 45.5 e | 23.0 c |
| | MSE | 11.146 | 266.91 | 225.18 | 6.143 | 194.81 |
| | p-value | < 0.0001 | 0.0007 | 0.0050 | < 0.0001 | < 0.0001 |
| | F-value | 70.89 | 5.19 | 3.91 | 237.60 | 13.31 |
| PM$_{2.5}$ (%) | | | | | | |
| | Unloading | 1.33 bc | 1.72 a | 0.66 a | 2.8 ab | 2.2 a |
| | # 1 Pull | 1.08 c | 0.19 b | 0.19 a | 1.9 b | 0.4 c |
| | # 2 Pull | 1.05 c | 0.81 ab | 0.87 a | 3.1 ab | 0.7 c |
| | #3A Pull | 1.02 c | 1.40 a | 0.78 a | 3.2 a | 1.9 ab |
| | #3B Pull | 2.29 a | 0.89 ab | 1.04 a | 2.6 ab | 0.8 bc |
| | Lint Basket Pull | 1.85 ab | 1.63 a | 0.88 a | 2.6 ab | 0.3 c |
| | MSE | 0.1895 | 0.696 | 0.458 | 0.834 | 0.636 |
| | p-value | < 0.0001 | 0.0022 | 0.1475 | 0.0432 | < 0.0001 |
| | F-value | 12.89 | 4.42 | 1.73 | 2.50 | 9.03 |

* MMD's reported in terms of aerodynamic equivalent diameter with a particle density of 1.96 g/cm$^3$.

**Table 41.** $PM_{10}$ source sampling component gravimetric weights for New Mexico Gin as determined by EEMC testing and PM percentages determined by Coulter Counter analyses.

| | | Cyclone Wash | | | Post-Cyclone Wash | | | Filter | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Exhaust | Run No. | Wt. (g) | $PM_{10}$ (%) | $PM_{2.5}$ (%) | Wt. (g) | $PM_{10}$ (%) | $PM_{2.5}$ (%) | Wt. (g) | $PM_{10}$ (%) | $PM_{2.5}$ (%) |
| Unloading | 1 | 0.0520 | 51.96 | 1.0142 | 0.0057 | 65.02 | 3.7499 | 0.0521 | 82.81 | 1.2629 |
| | 2 | 0.0098 | 29.76 | 0.0892 | 0.0020 | 42.10 | 1.0907 | 0.0764 | 84.12 | 1.5834 |
| | 3 | 0.0580 | 54.49 | 0.8511 | 0.0043 | 78.83 | 0.3298 | 0.0660 | 83.32 | 1.1650 |
| # 1 Pull | 1 | 0.0067 | 56.54 | 2.0736 | 0.0025 | 64.98 | 0.9832 | 0.0171 | 75.86 | 1.2075 |
| | 2 | 0.0063 | 21.98 | 0.2589 | 0.0009 | 66.81 | 0.7931 | 0.0186 | 81.11 | 0.8654 |
| | 3 | 0.0082 | 26.01 | 0.2872 | 0.0028 | 64.82 | 0.6569 | 0.0171 | 80.10 | 1.0795 |
| # 2 Pull | 1 | 0.0482 | 18.86 | 0.3146 | 0.0016 | 9.38 | 0.0007 | 0.0783 | 86.64 | 1.0219 |
| | 2 | 0.0291 | 12.17 | 0.0516 | 0.0028 | 77.57 | 0.9113 | 0.0663 | 85.53 | 0.9956 |
| | 3 | 0.0427 | 14.25 | 0.2020 | 0.0073 | 13.19 | 0.0222 | 0.0725 | 83.08 | 1.1834 |
| #3A Pull | 1 | 0.0017 | 45.82 | 1.1637 | 0.0052 | 39.30 | 1.5060 | 0.0036 | 70.00 | 1.1400 |
| | 2 | 0.0077 | 28.19 | 0.3860 | 0.0002 | 40.11 | 1.3225 | 0.0077 | 77.63 | 0.9417 |
| | 3 | 0.0035 | 36.99 | 0.8353 | 0.0037 | 60.91 | 1.3621 | 0.0083 | 76.76 | 0.9421 |
| #3B Pull | 1 | 0.0204 | 48.84 | 2.2396 | 0.0017 | 58.91 | 1.1884 | 0.0171 | 63.52 | 2.6381 |
| | 2 | 0.0149 | 40.27 | 0.8901 | 0.0013 | 30.47 | 0.2308 | 0.0194 | 61.70 | 1.4386 |
| | 3 | 0.0194 | 9.06 | 0.0047 | 0.0010 | 56.07 | 1.2312 | 0.0213 | 64.77 | 2.7830 |
| Lint Basket Pull | 1 | 0.0212 | 36.11 | 1.0189 | 0.0021 | 54.61 | 1.2899 | 0.0130 | 59.43 | 2.8640 |
| | 2 | 0.0008 | 55.79 | 1.6073 | 0.0040 | 29.43 | 0.4515 | 0.0106 | 72.05 | 1.2654 |
| | 3 | 0.0238 | 6.61 | 0.0023 | 0.0008 | 51.18 | 3.0952 | 0.0075 | 61.73 | 1.4133 |

**Table 42.** TSP source sampling component gravimetric weights for Mesa Gin as determined by EEMC testing and PM percentages determined by Coulter Counter analyses.

| | | Cyclone Wash | | | Filter | | |
|---|---|---|---|---|---|---|---|
| Exhaust | Run No. | Wt. (g) | $PM_{10}$ (%) | $PM_{2.5}$ (%) | Wt. (g) | $PM_{10}$ (%) | $PM_{2.5}$ (%) |
| Unloading | 1 | 0.0081 | 53.45 | 4.4290 | 0.0464 | 53.05 | 1.0859 |
| | 2 | 0.0776 | 68.70 | 1.1055 | 0.1021 | 52.50 | 3.4464 |
| | 3 | 0.0657 | 68.13 | 1.0865 | 0.1319 | 54.75 | 3.9585 |
| # 1 Pull | 1 | 0.0098 | 9.53 | 0.0088 | 0.0171 | 66.90 | 2.8849 |
| | 2 | 0.0248 | 60.65 | 0.9505 | 0.0329 | 66.12 | 2.8581 |
| | 3 | 0.0070 | 64.36 | 1.1376 | 0.0239 | 64.20 | 3.5673 |
| # 2 Pull | 1 | 0.0414 | 41.53 | 0.6145 | 0.1118 | 75.76 | 1.7408 |
| | 2 | 0.0354 | 19.53 | 0.3482 | 0.1253 | 76.17 | 1.7090 |
| | 3 | 0.0473 | 16.32 | 0.3136 | 0.1071 | 73.67 | 2.1780 |
| #3A Pull | 1 | 0.0102 | 56.32 | 1.7662 | 0.0039 | 67.06 | 4.2903 |
| | 2 | 0.0067 | 52.74 | 2.6918 | 0.0072 | 73.37 | 2.5950 |
| | 3 | 0.0033 | 70.85 | 1.1760 | 0.0076 | 70.04 | 2.6702 |
| #3B Pull | 1 | 0.0210 | 50.14 | 0.3864 | 0.0238 | 46.58 | 2.2517 |
| | 2 | 0.0266 | 45.70 | 0.7645 | 0.0298 | 49.23 | 3.4565 |
| | 3 | 0.0166 | 55.29 | 1.2333 | 0.0497 | 42.32 | 1.9821 |
| Lint Basket Pull | 1 | 0.0157 | 40.80 | 0.7819 | 0.0242 | 47.07 | 2.5351 |
| | 2 | 0.0066 | 18.83 | 0.0448 | 0.0287 | 49.07 | 4.2122 |
| | 3 | 0.0126 | 9.44 | 0.0057 | 0.0288 | 40.62 | 1.0372 |

**Table 43.** Mesa Gin exhaust emission factors as determined by Method 5 source sampling and Coulter Counter particle size analyses.

| | | Source Sampling | Coulter Counter Analysis (True) | | | |
|---|---|---|---|---|---|---|
| Exhaust | Run No. | TSP** | $PM_{10}$** | $PM_{2.5}$** | $PM_{10}$/TSP Ratio | $PM_{2.5}$/TSP Ratio |
| Unloading | 1 | 0.0534 (0.1177) | 0.0284 (0.0625) | 0.0008 (0.0019) | 53.1 | 1.58 |
| | 2 | 0.1774 (0.3912) | 0.1056 (0.2327) | 0.0043 (0.0095) | 59.5 | 2.44 |
| | 3 | 0.1943 (0.4283) | 0.1150 (0.2536 | 0.0058 (0.0129) | 59.2 | 3.00 |
| | Average | 0.1417 (0.3124) a | 0.0830 (0.1829) a | 0.0037 (0.0081) a | 57.3 a | 2.34 a |
| # 1 Pull | 1 | 0.1205 (0.2657) | 0.0802 (0.1767) | 0.0017 (0.0038) | 66.5 | 1.44 |
| | 2 | 0.1275 (0.2811) | 0.0812 (0.1791) | 0.0018 (0.0040) | 63.7 | 1.41 |
| | 3 | 0.1217 (0.2684) | 0.0683 (0.1506) | 0.0020 (0.0043) | 56.1 | 1.61 |
| | Average | 0.1233 (0.2717) b | 0.0766 (0.1688) b | 0.0018 (0.0040) b | 62.1 a | 1.48 a |
| # 2 Pull | 1 | 0.0163 (0.0360) | 0.0075 (0.0165) | 0.0003 (0.0007) | 46.0 | 1.84 |
| | 2 | 0.0350 (0.0772) | 0.0223 (0.0492) | 0.0007 (0.0016) | 63.8 | 2.04 |
| | 3 | 0.0188 (0.0414) | 0.0121 (0.0266) | 0.0006 (0.0012) | 64.2 | 3.02 |
| | Average | 0.0234 (0.0515) a | 0.0140 (0.0308) a | 0.0005 (0.0012) ab | 58.0 a | 2.30 a |
| #3A Pull | 1 | 0.0046 (0.0101) | 0.0027 (0.0060) | 0.0001 (0.0002) | 59.3 | 2.46 |
| | 2 | 0.0045 (0.0100) | 0.0029 (0.0063) | 0.0001 (0.0003) | 63.4 | 2.64 |
| | 3 | 0.0036 (0.0078) | 0.0025 (0.0055) | 0.0001 (0.0002) | 70.3 | 2.22 |
| | Average | 0.0042 (0.0093) b | 0.0027 (0.0059) b | 0.0001 (0.0002) b | 64.3 a | 2.44 a |
| #3B Pull | 1 | 0.0136 (0.0299) | 0.0065 (0.0144) | 0.0002 (0.0004) | 48.2 | 1.38 |
| | 2 | 0.0171 (0.0377) | 0.0081 (0.0179) | 0.0004 (0.0008) | 47.6 | 2.19 |
| | 3 | 0.0201 (0.0444) | 0.0092 (0.0202) | 0.0004 (0.0008) | 45.6 | 1.79 |
| | Average | 0.0169 (0.0373) b | 0.0080 (0.0175) b | 0.0003 (0.0007) b | 47.1 ab | 1.79 a |
| Lint Basket Pull | 1 | 0.0252 (0.0556) | 0.0113 (0.0248) | 0.0005 (0.0010) | 44.6 | 1.85 |
| | 2 | 0.0224 (0.0493) | 0.0097 (0.0214) | 0.0008 (0.0017) | 43.4 | 3.43 |
| | 3 | 0.0262 (0.0577) | 0.0081 (0.0180) | 0.0002 (0.0004) | 31.1 | 0.72 |
| | Average | 0.0246 (0.0542) b | 0.0097 (0.0214) b | 0.0005 (0.0011) b | 39.7 b | 2.00a |
| MSE | | 0.0049 | 0.0019 | 0.0005 | 0.004 | 0.00005 |
| p-value*** | | 0.0004 | 0.0005 | 0.0099 | 0.0033 | 0.5506 |
| F-value | | 10.78 | 10.38 | 5.08 | 6.75 | 0.83 |

* Emission factors correspond only to the one cyclone tested and not the entire system.

** TSP, $PM_{10}$, and $PM_{2.5}$ emission factors reported in kg of PM emitted per bale of lint cotton produced (lb of PM emitted per bale of lint cotton produced).

*** p-value is the test of significance of means being equal in the PROC Mixed procedure of SAS.

Means in the same column followed by the same letter are not statistically different at the 0.05 level of siginificance.

**Table 44.** Mesa Gin exhaust emission factors as determined by Method 201a source sampling and Coulter Counter particle size analyses.

| | | Source Sampling | | | Coulter Counter Analysis (True) | | | |
|---|---|---|---|---|---|---|---|---|
| Exhaust | Run No. | TSP$^{**}$ | PM$_{10}^{**}$ | PM$_{10}$/TSP Ratio | PM$_{10}^{**}$ | PM$_{2.5}^{**}$ | PM$_{10}$/TSP Ratio | PM$_{2.5}$/TSP Ratio |
| Unloading | 1 | 0.1642 | 0.0864 | 52.6 | 0.1105 | 0.0046 | 67.3 | 1.27 |
| | | (0.3620) | (0.1906) | | (0.2436) | (0.0021) | | |
| | 2 | 0.1204 | 0.1070 | 88.9 | 0.0929 | 0.0017 | 77.1 | 1.41 |
| | | (0.2654) | (0.2359) | | (0.2047) | (0.0037) | | |
| | 3 | 0.1763 | 0.0966 | 54.8 | 0.1236 | 0.0018 | 70.1 | 1.00 |
| | | (0.3886) | (0.2129) | | (0.2725) | (0.0039) | | |
| | Average | 0.1536 | 0.0967 | 65.4 a | 0.1090 | 0.0018 | 71.5 a | 1.23 a |
| | | (0.3387) a | (0.2131) a | | (0.2403) a | (0.0041) a | | |
| # 1 Pull | 1 | 0.1445 | 0.0901 | 62.4 | 0.0870 | 0.0011 | 60.2 | 0.74 |
| | | (0.3186) | (0.1987) | | (0.1917) | (0.0024) | | |
| | 2 | 0.1117 | 0.0786 | 70.4 | 0.0710 | 0.0008 | 63.6 | 0.71 |
| | | (0.2462) | (0.1733) | | (0.1565) | (0.0018) | | |
| | 3 | 0.1407 | 0.0916 | 65.1 | 0.0773 | 0.0011 | 54.9 | 0.77 |
| | | (0.3101) | (0.2020) | | (0.1703) | (0.0024) | | |
| | Average | 0.1323 | 0.0868 | 66.0 a | 0.0784 | 0.0010 | 59.6 ab | 0.74 a |
| | | (0.2917) a | (0.1913) a | | (0.1729) b | (0.0022) b | | |
| # 2 Pull | 1 | 0.0235 | 0.0175 | 74.5 | 0.0164 | 0.0003 | 69.9 | 1.41 |
| | | (0.0517) | (0.0385) | | (0.0361) | (0.0007) | | |
| | 2 | 0.0230 | 0.0174 | 75.6 | 0.0152 | 0.0002 | 66.2 | 0.71 |
| | | (0.0508) | (0.0384) | | (0.0336) | (0.0004) | | |
| | 3 | 0.0255 | 0.0181 | 70.8 | 0.0160 | 0.0002 | 62.8 | 0.81 |
| | | (0.0563) | (0.0399) | | (0.0354) | (0.0005) | | |
| | Average | 0.0240 | 0.0177 | 73.6 a | 0.0159 | 0.0002 | 66.3 ab | 0.98 a |
| | | (0.0529) b | (0.0389) b | | (0.0350) c | (0.0005) c | | |
| #3A Pull | 1 | 0.0044 | 0.0037 | 83.8 | 0.0022 | 0.0001 | 50.9 | 1.33 |
| | | (0.0097) | (0.0081) | | (0.0049) | (0.0001) | | |
| | 2 | 0.0065 | 0.0033 | 50.6 | 0.0034 | 0.0000 | 52.7 | 0.67 |
| | | (0.0143) | (0.0072) | | (0.0075) | (0.0001) | | |
| | 3 | 0.0065 | 0.0050 | 77.4 | 0.0042 | 0.0001 | 64.0 | 1.34 |
| | | (0.0143) | (0.0111) | | (0.0092) | (0.0002) | | |
| | Average | 0.0058 | 0.0040 | 70.6 a | 0.0033 | 0.0001 | 55.9 ab | 1.11 a |
| | | (0.0128) b | (0.0088) b | | (0.0072) c | (0.0001) c | | |

**Table 44 (cont.).**

| | | Source Sampling | | | Coulter Counter Analysis (True) | | | |
|---|---|---|---|---|---|---|---|---|
| Exhaust | Run No. | $TSP^{**}$ | $PM_{10}^{**}$ | $PM_{10}$/TSP Ratio | $PM_{10}^{**}$ | $PM_{2.5}^{**}$ | $PM_{10}$/TSP Ratio | $PM_{2.5}$/TSP Ratio |
| #3B Pull | 1 | 0.0190 | 0.0091 | 48.0 | 0.0106 | 0.0003 | 55.7 | 1.67 |
| | | (0.0419) | (0.0201) | | (0.0233) | (0.0007) | | |
| | 2 | 0.0180 | 0.0105 | 58.1 | 0.0093 | 0.0001 | 51.6 | 0.79 |
| | | (0.0397) | (0.0231) | | (0.0205) | (0.0003) | | |
| | 3 | 0.0211 | 0.0113 | 53.5 | 0.0081 | 0.0004 | 38.6 | 1.93 |
| | | (0.0465) | (0.0248) | | (0.0180) | (0.0009) | | |
| | Average | 0.0194 | 0.0103 | 53.2 a | 0.0093 | 0.0003 | 48.6 ab | 1.46 a |
| | | (0.0427) b | (0.0227) b | | (0.0206) c | (0.0006) c | | |
| Lint Basket | 1 | 0.0328 | 0.0136 | 41.6 | 0.0149 | 0.0007 | 45.5 | 2.04 |
| | | (0.0722) | (0.0300) | | (0.0329) | (0.0015) | | |
| Pull | 2 | 0.0140 | 0.0133 | 94.8 | 0.0084 | 0.0001 | 60.1 | 0.99 |
| | | (0.0309) | (0.0293) | | (0.0186) | (0.0003) | | |
| | 3 | 0.0292 | 0.0075 | 25.9 | 0.0060 | 0.0001 | 20.6 | 0.41 |
| | | (0.0644) | (0.0166) | | (0.0133) | (0.0003) | | |
| | Average | 0.0253 | 0.0115 | 54.1 a | 0.0098 | 0.0003 | 42.1 b | 1.14 a |
| | | (0.0558) b | (0.0253) b | | (0.0216) c | (0.0007) c | | |
| MSE | | 0.0011 | 0.0001 | 0.0346 | 0.0003 | 1.59 e-7 | 0.0098 | 0.00002 |
| p-value$^{***}$ | | < 0.0001 | < 0.0001 | 0.6878 | < 0.0001 | < 0.0001 | 0.0304 | 0.5909 |
| F-value | | 58.78 | 187.01 | 0.62 | 109.96 | 42.37 | 3.66 | 0.77 |

* Emission factors correspond only to the one cyclone tested and not the entire system.

** TSP, $PM_{10}$, and $PM_{2.5}$ emission factors reported in kg of PM emitted per bale of lint cotton produced (lb of PM emitted per bale of lint cotton produced).

*** p-value is the test of significance of means being equal in the PROC Mixed procedure of SAS.

Means in the same column followed by the same letter are not statistically different at the 0.05 level of significance.

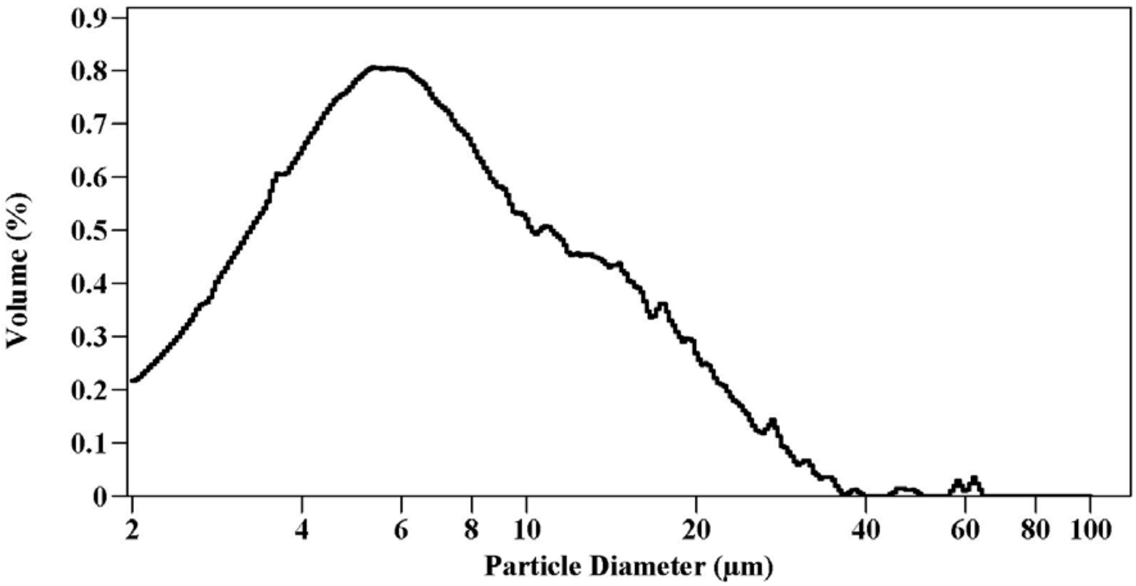

**Figure 150.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 5 glass fiber filter obtained from stack sampling conducted on a roller gin unloading system exhaust.

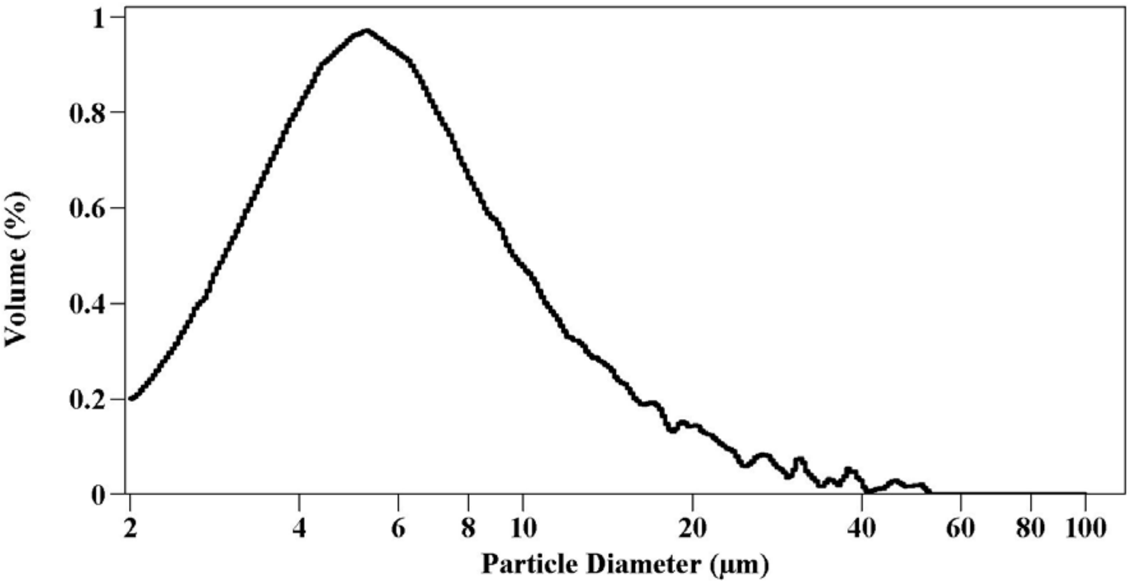

**Figure 151.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 5 stack sampling conducted on a roller gin unloading system exhaust.

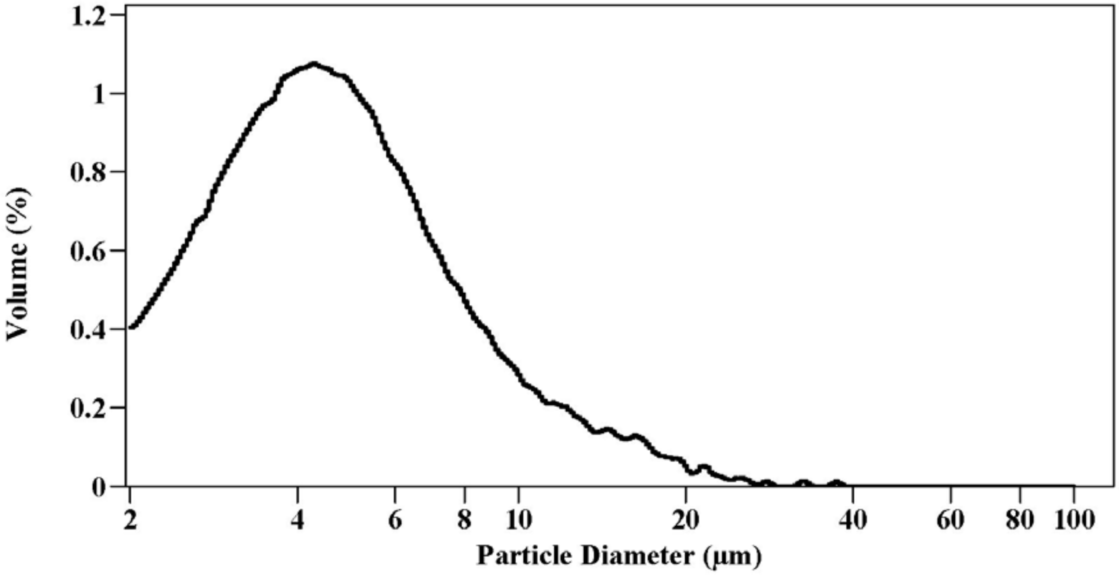

**Figure 152.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 5 glass fiber filter obtained from stack sampling conducted on a roller gin 1st pre-cleaning system exhaust.

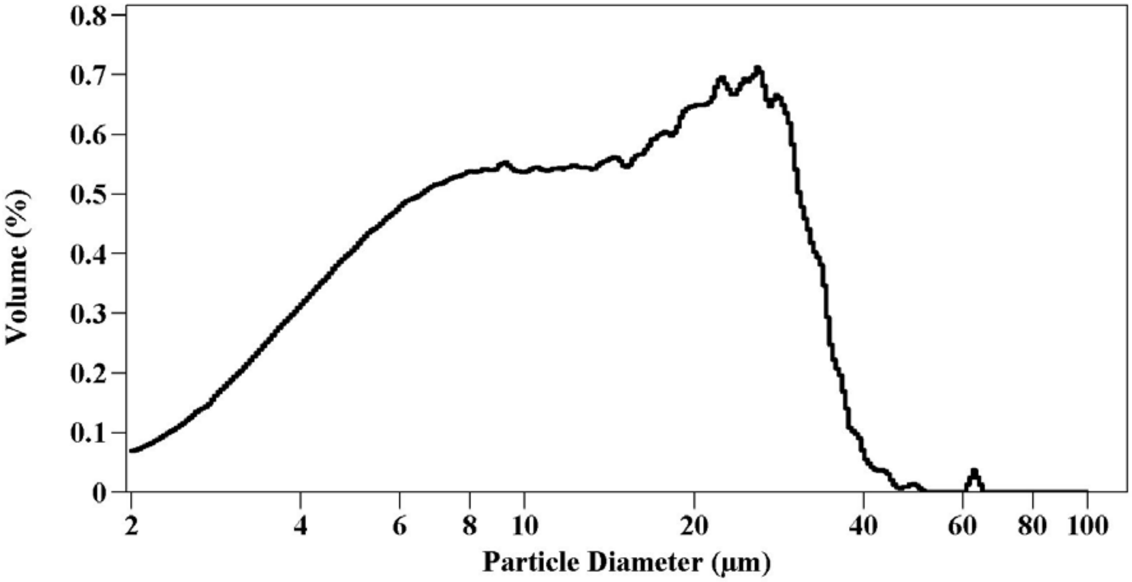

**Figure 153.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 5 stack sampling conducted on a roller gin 1st pre-cleaning system exhaust.

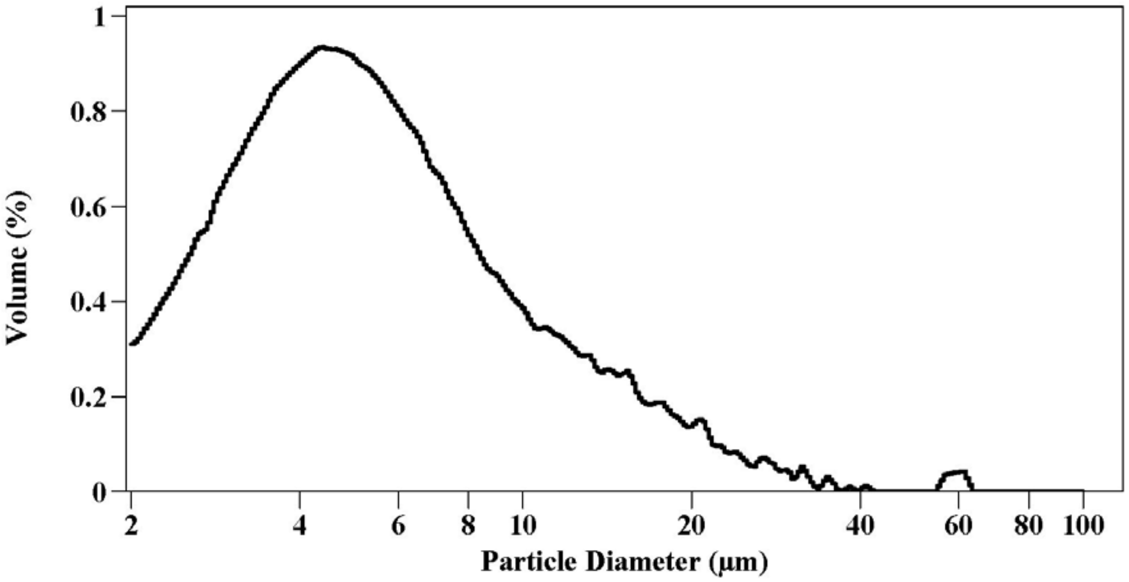

**Figure 154.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 5 glass fiber filter obtained from stack sampling conducted on a roller gin 2nd pre-cleaning system exhaust.

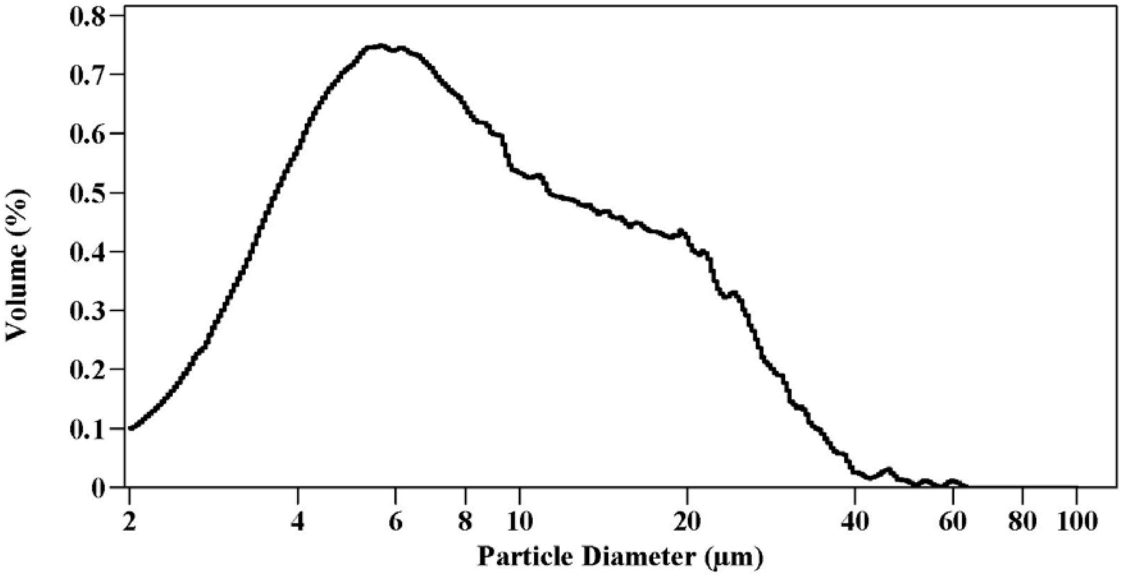

**Figure 155.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 5 stack sampling conducted on a roller gin 2nd pre-cleaning system exhaust.

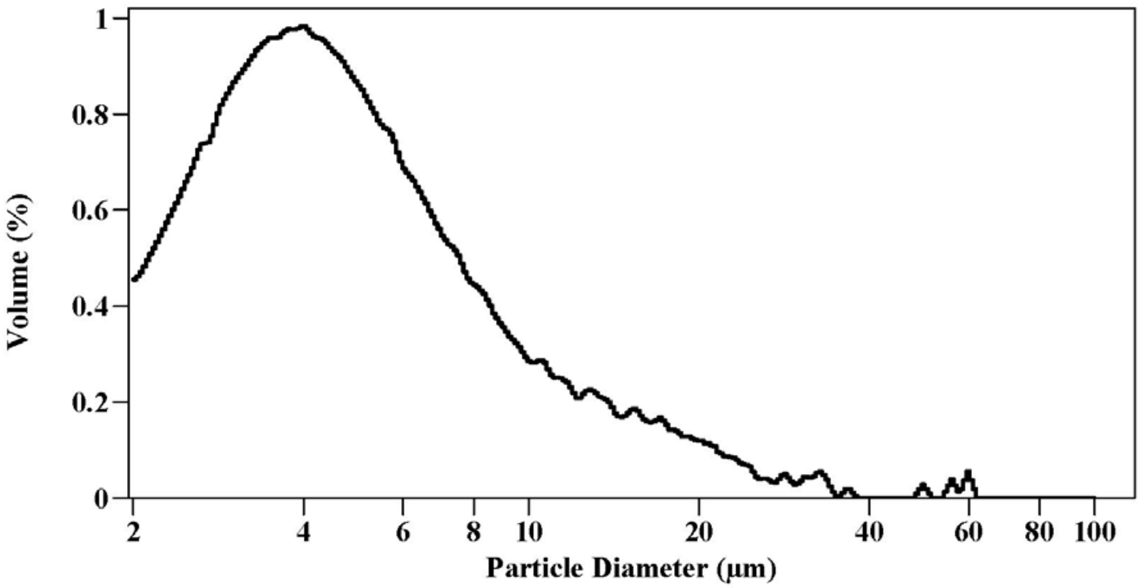

**Figure 156.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 5 glass fiber filter obtained from stack sampling conducted on a roller gin 3rd incline cleaner (system A) exhaust.

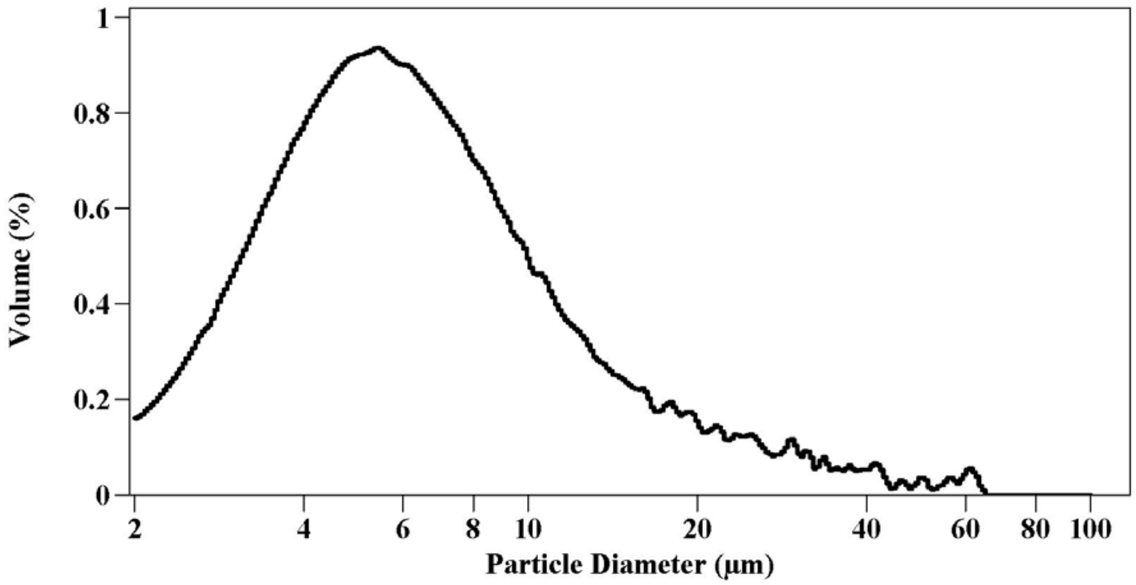

**Figure 157.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 5 stack sampling conducted on a roller gin 3rd incline cleaner (system A) exhaust.

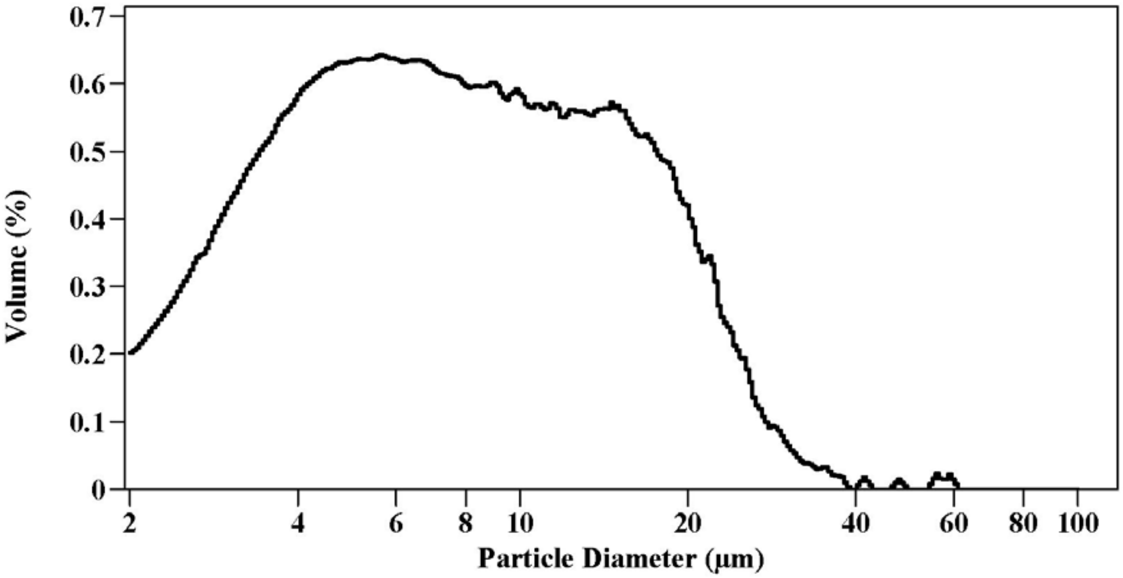

**Figure 158.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 5 glass fiber filter obtained from stack sampling conducted on a roller gin 3rd incline cleaner (system B) exhaust.

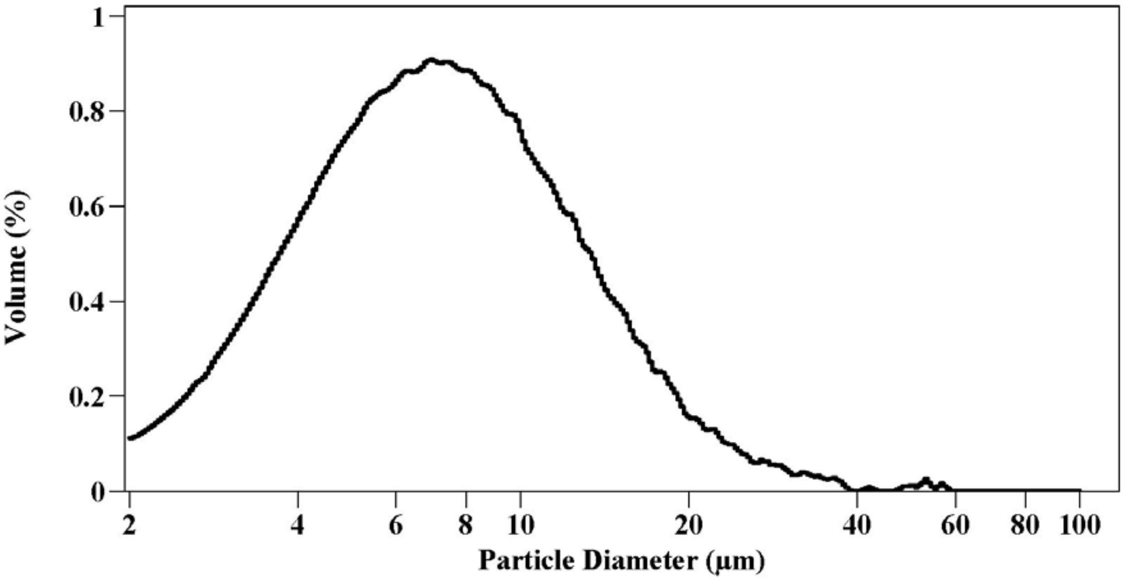

**Figure 159.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 5 stack sampling conducted on a roller gin 3rd incline cleaner (system B) exhaust.

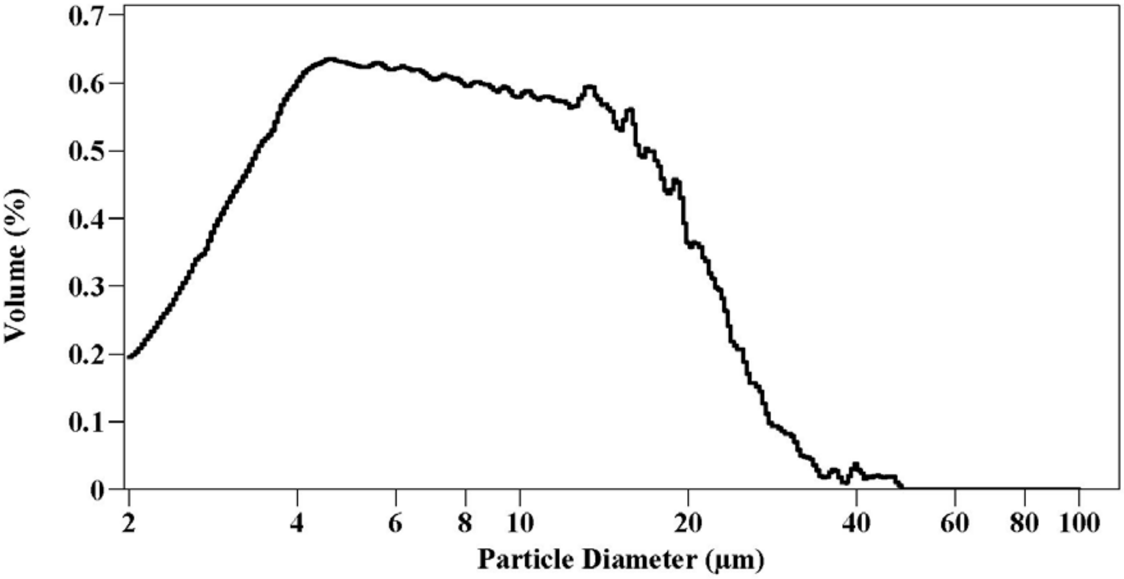

**Figure 160.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 5 glass fiber filter obtained from stack sampling conducted on a roller gin lint basket pull system exhaust.

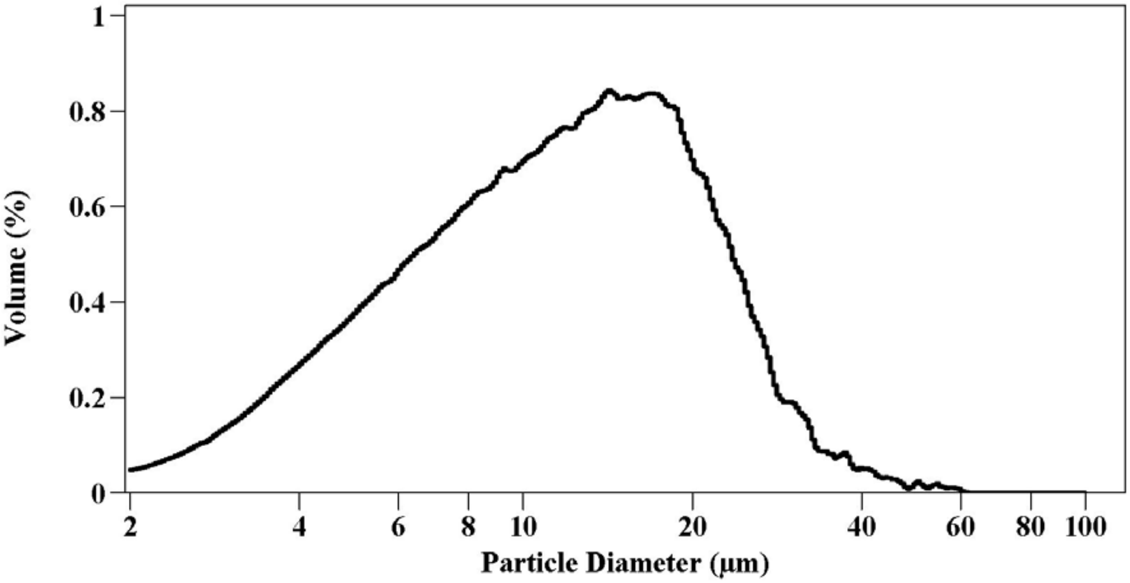

**Figure 161.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 5 stack sampling conducted on a roller gin lint basket pull system exhaust.

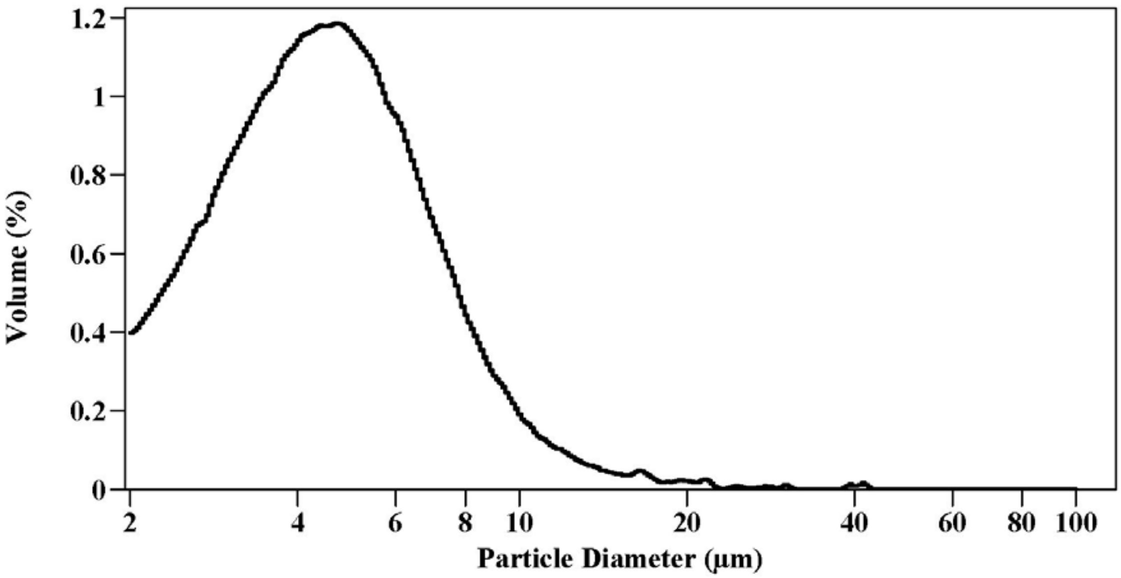

**Figure 162.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 201a glass fiber filter obtained from stack sampling conducted on a roller gin unloading system exhaust.

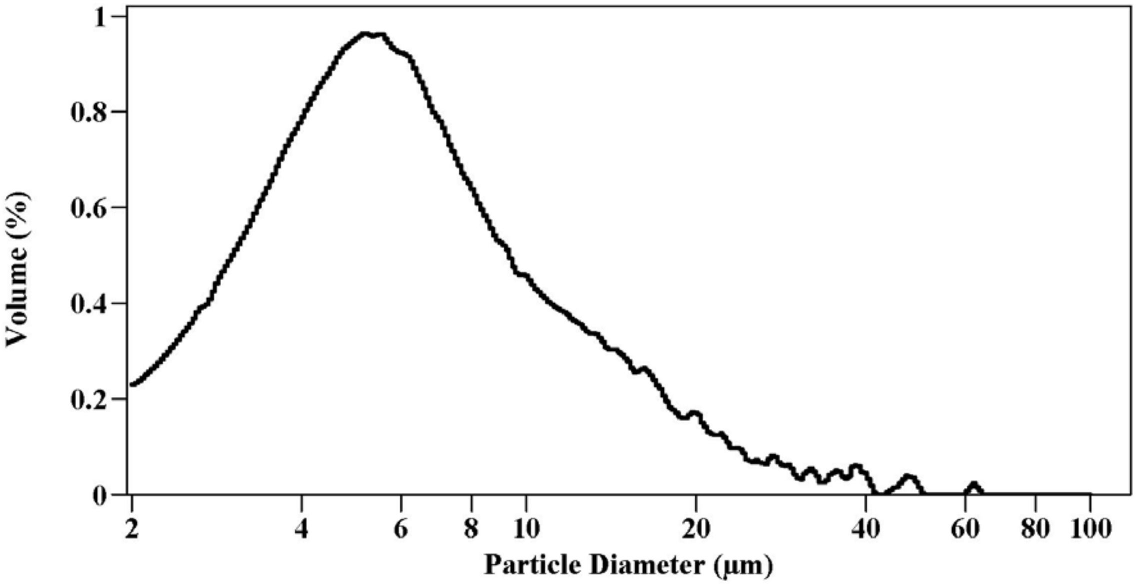

**Figure 163.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the post-cyclone wash obtained from Method 201a stack sampling conducted on a roller gin unloading system exhaust.

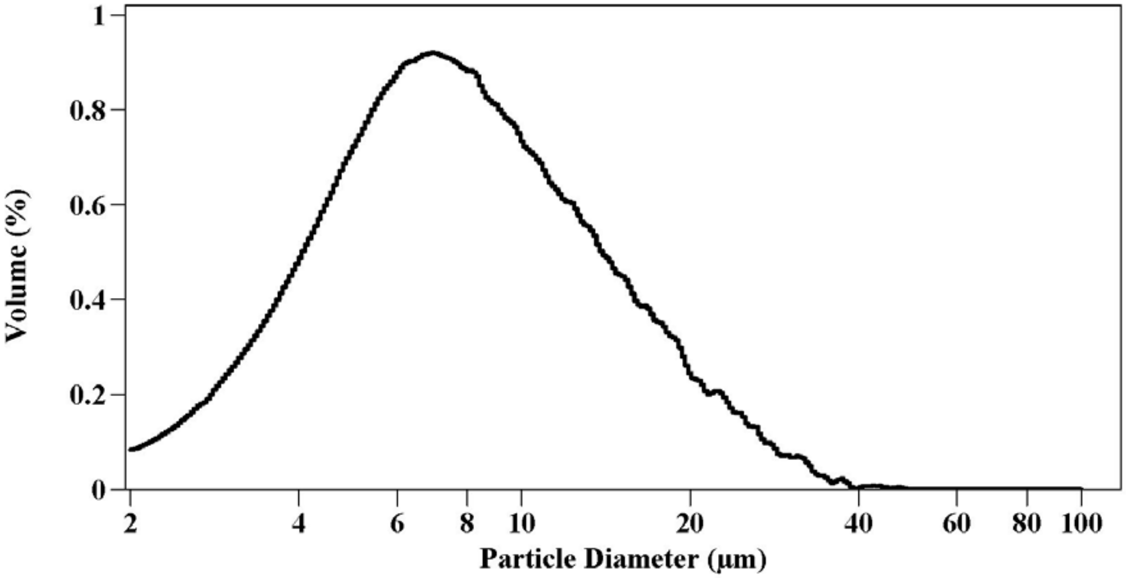

**Figure 164.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 201a stack sampling conducted on a roller gin unloading system exhaust.

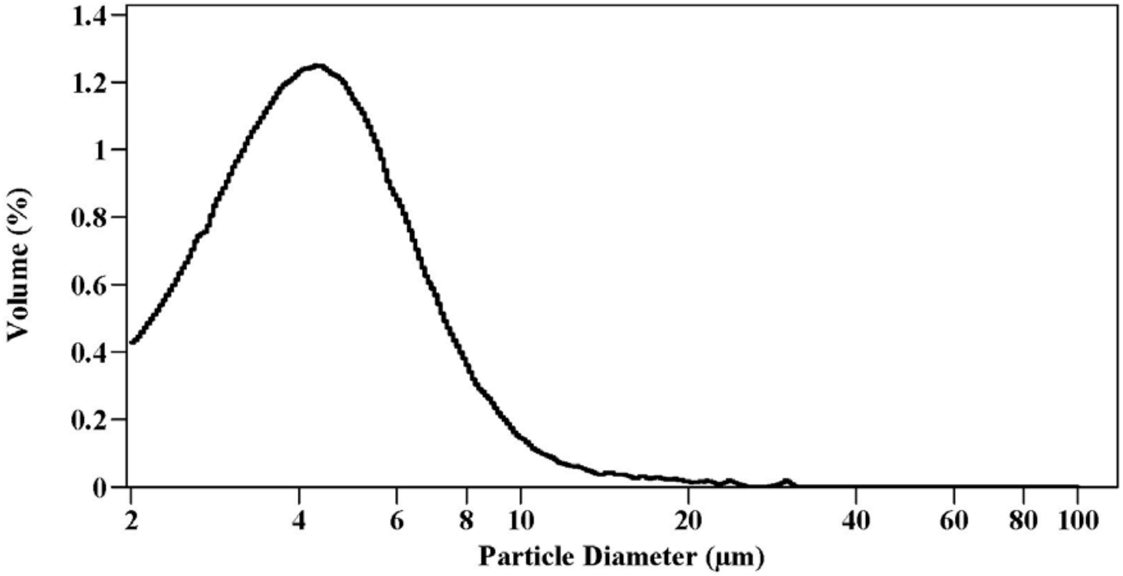

**Figure 165.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 201a glass fiber filter obtained from stack sampling conducted on a roller gin 1st pre-cleaning system exhaust.

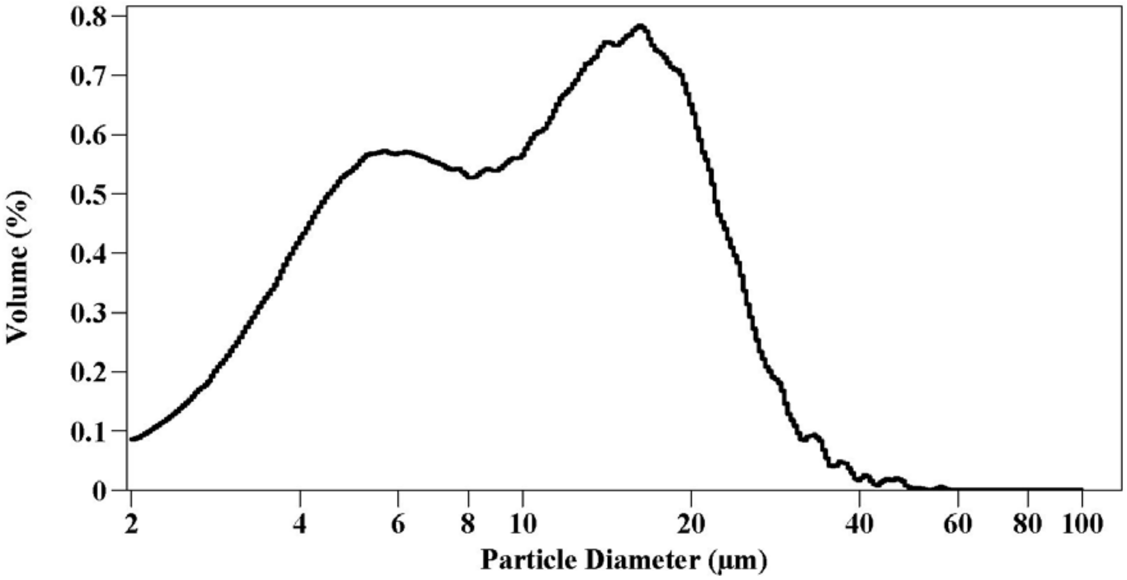

**Figure 166.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the post-cyclone wash obtained from Method 201a stack sampling conducted on a roller gin 1st pre-cleaning system exhaust.

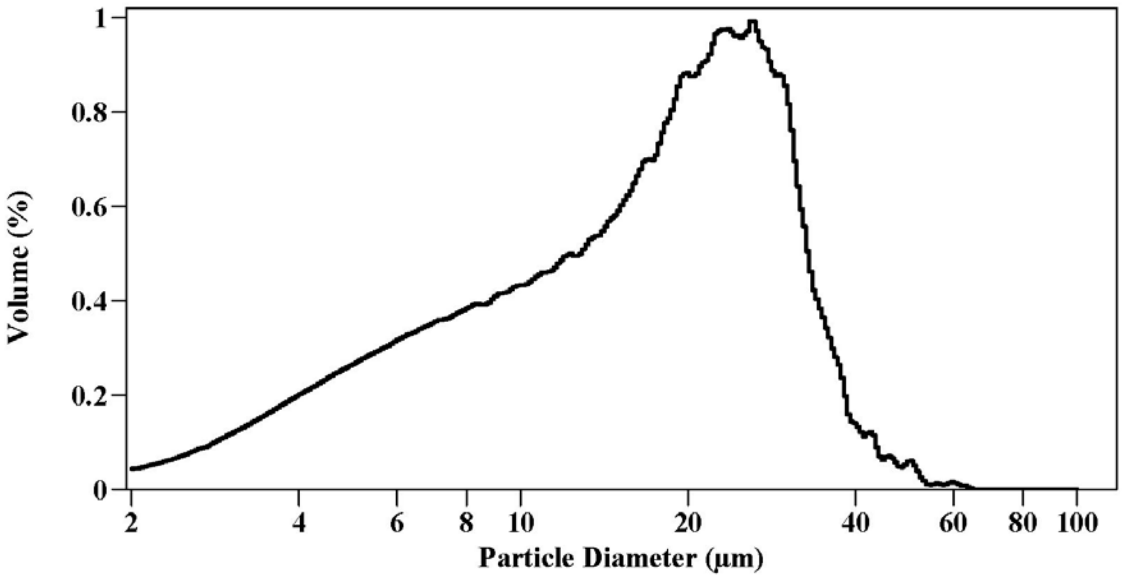

**Figure 167.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 201a stack sampling conducted on a roller gin 1st pre-cleaning system exhaust.

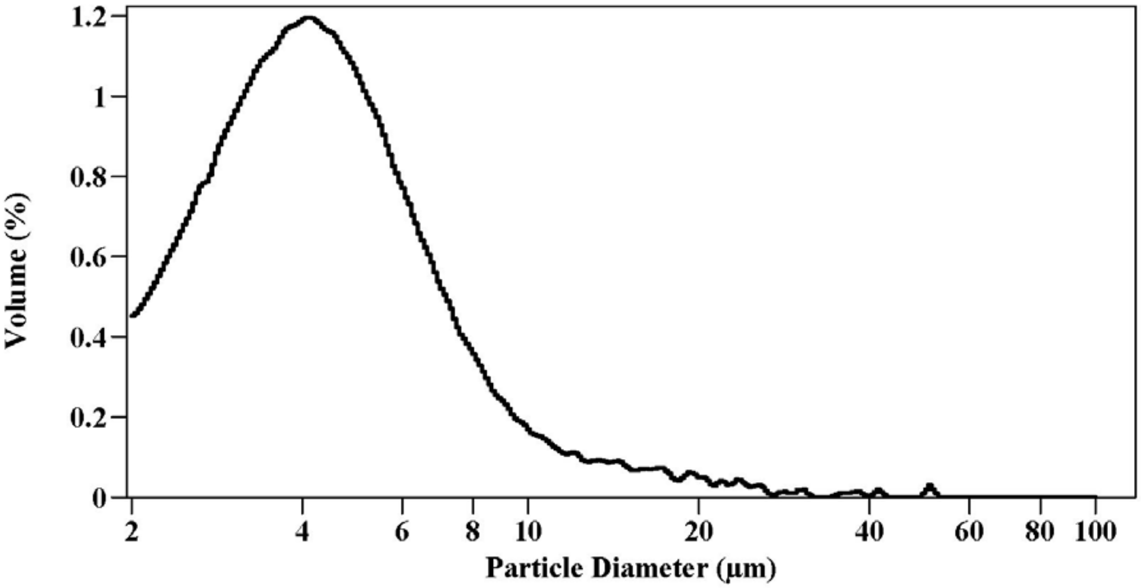

**Figure 168.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 201a glass fiber filter obtained from stack sampling conducted on a roller gin 2nd pre-cleaning system exhaust.

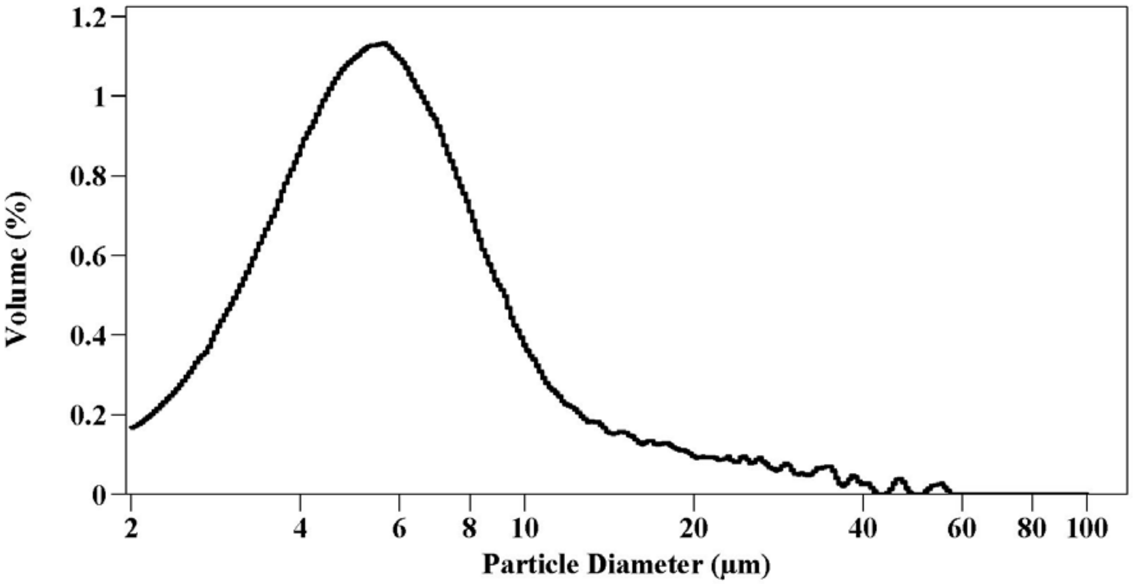

**Figure 169.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the post-cyclone wash obtained from Method 201a stack sampling conducted on a roller gin 2nd pre-cleaning system exhaust.

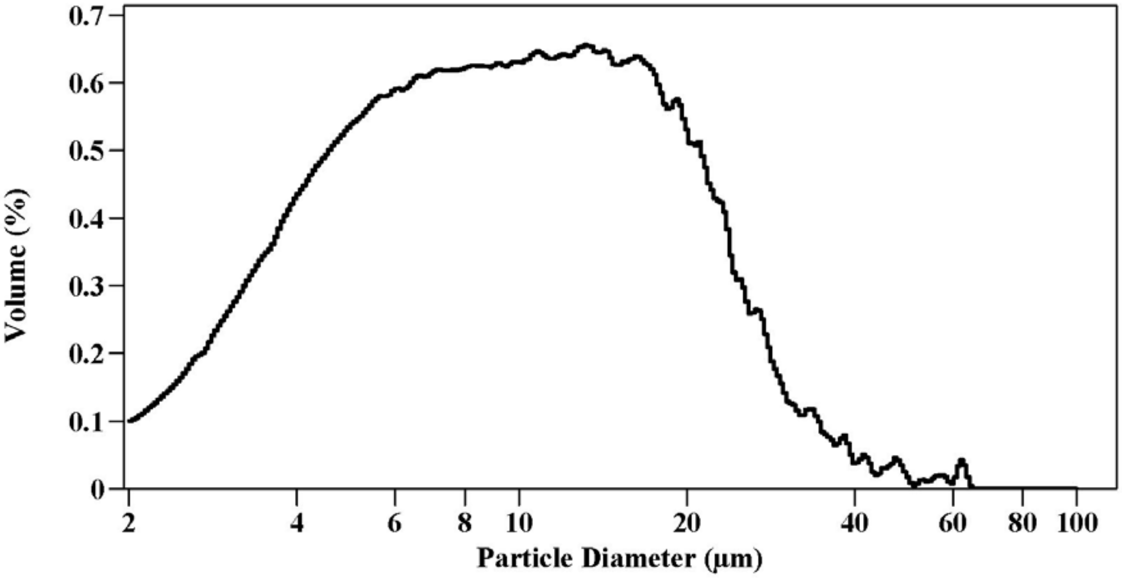

**Figure 170.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 201a stack sampling conducted on a roller gin 2nd pre-cleaning system exhaust.

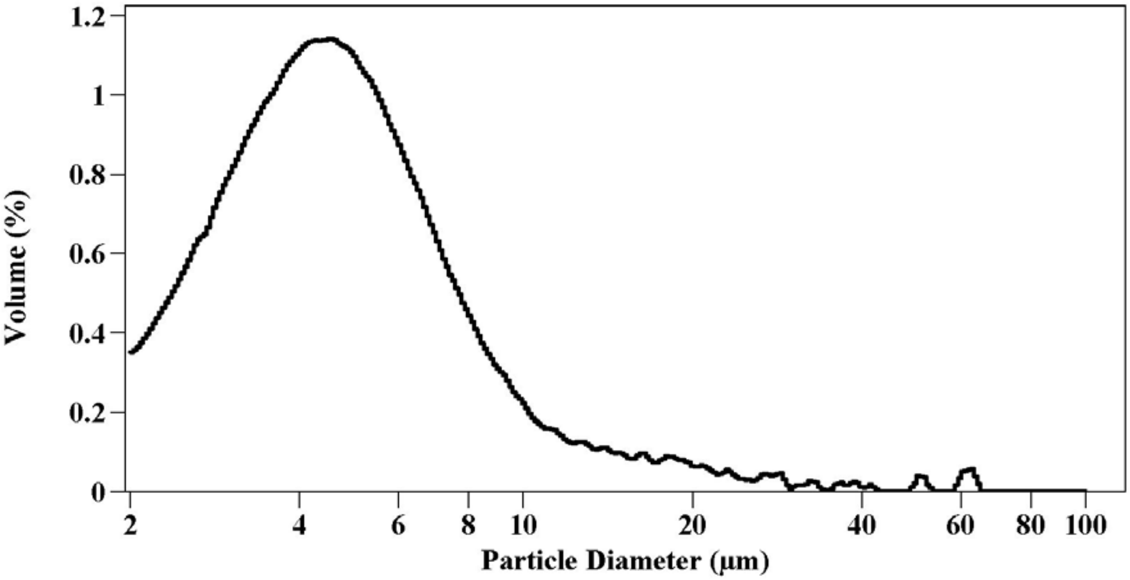

**Figure 171.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 201a glass fiber filter obtained from stack sampling conducted on a roller gin 3rd incline cleaner (system A) exhaust.

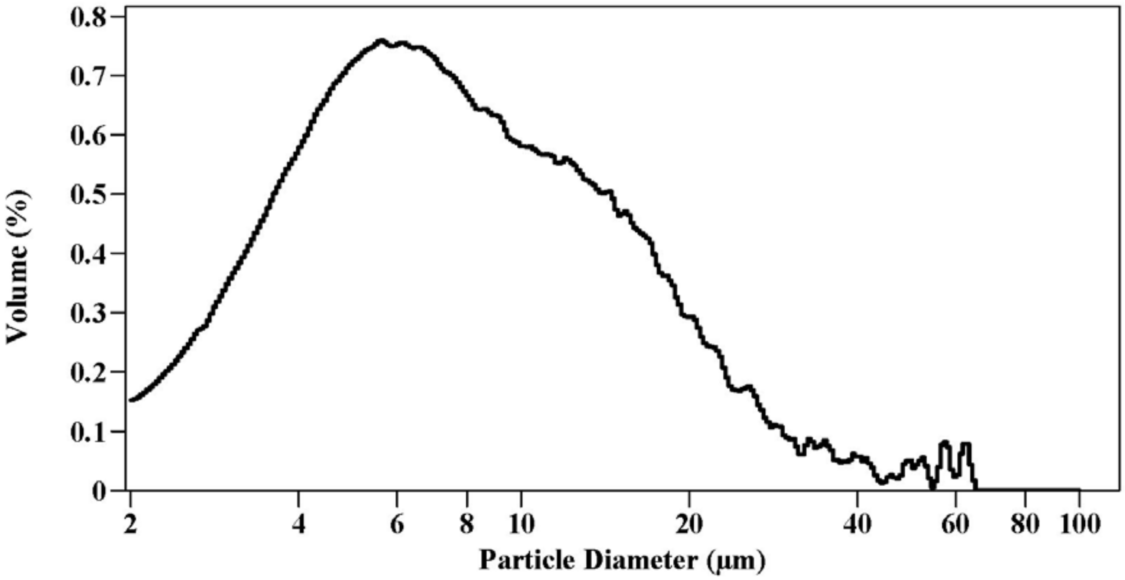

**Figure 172.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the post-cyclone wash obtained from Method 201a stack sampling conducted on a roller gin 3rd incline cleaner (system A) exhaust.

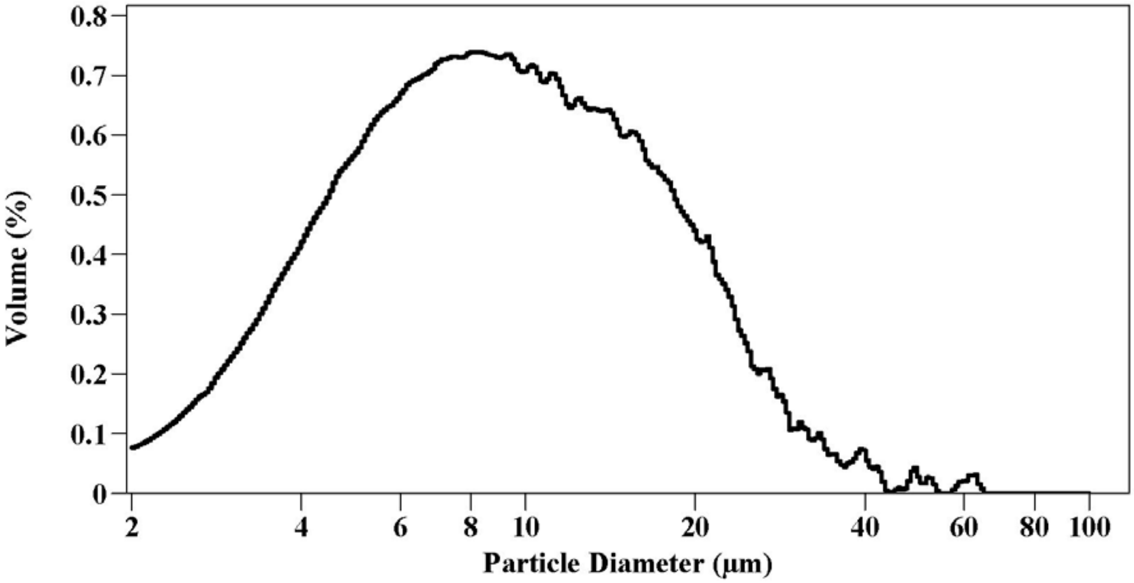

**Figure 173.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 201a stack sampling conducted on a roller gin 3rd incline cleaner (system A) exhaust.

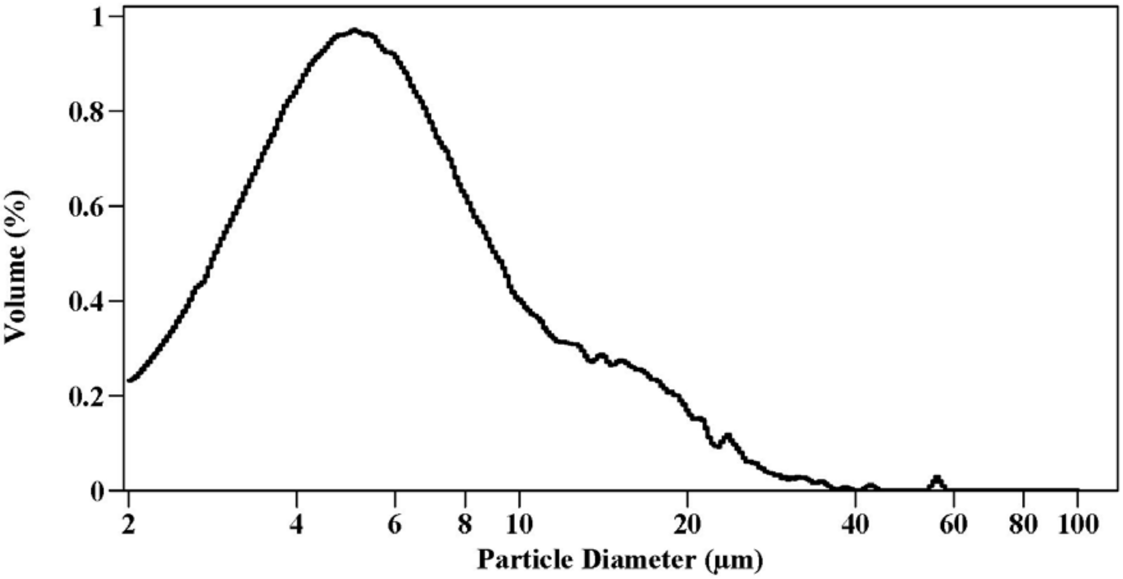

**Figure 174.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 201a glass fiber filter obtained from stack sampling conducted on a roller gin 3rd incline cleaner (system B) exhaust.

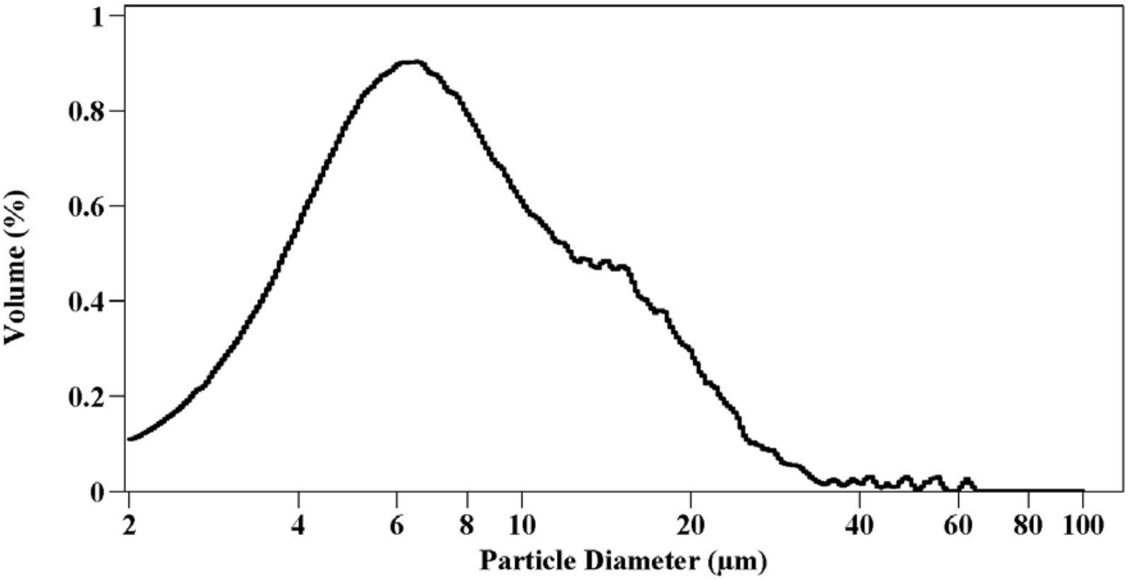

**Figure 175.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the post-cyclone wash obtained from Method 201a stack sampling conducted on a roller gin 3rd incline cleaner (system B) exhaust.

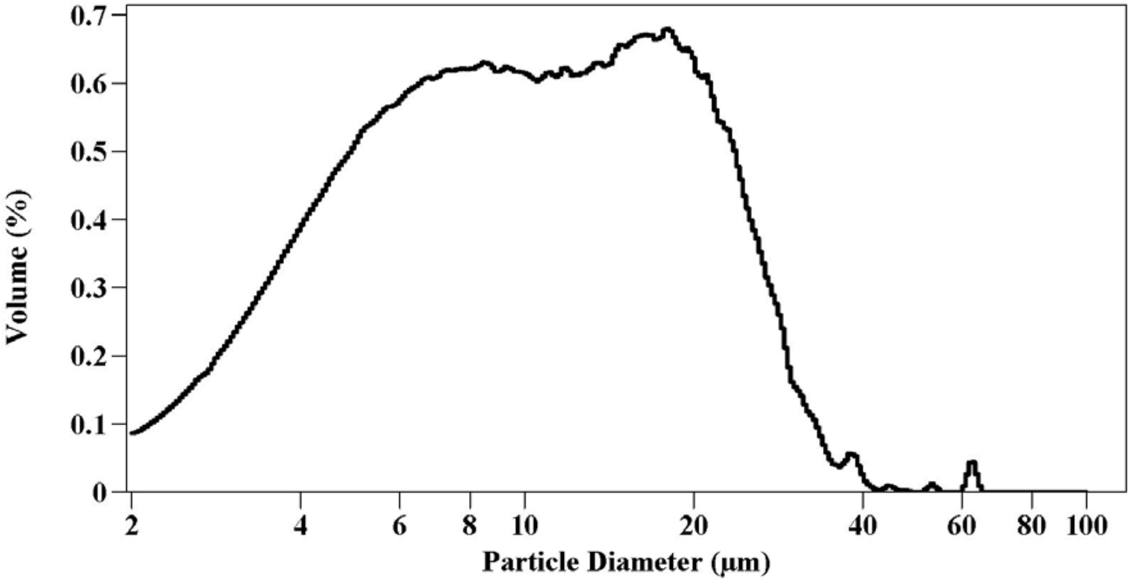

**Figure 176.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 201a stack sampling conducted on a roller gin 3rd incline cleaner (system B) exhaust.

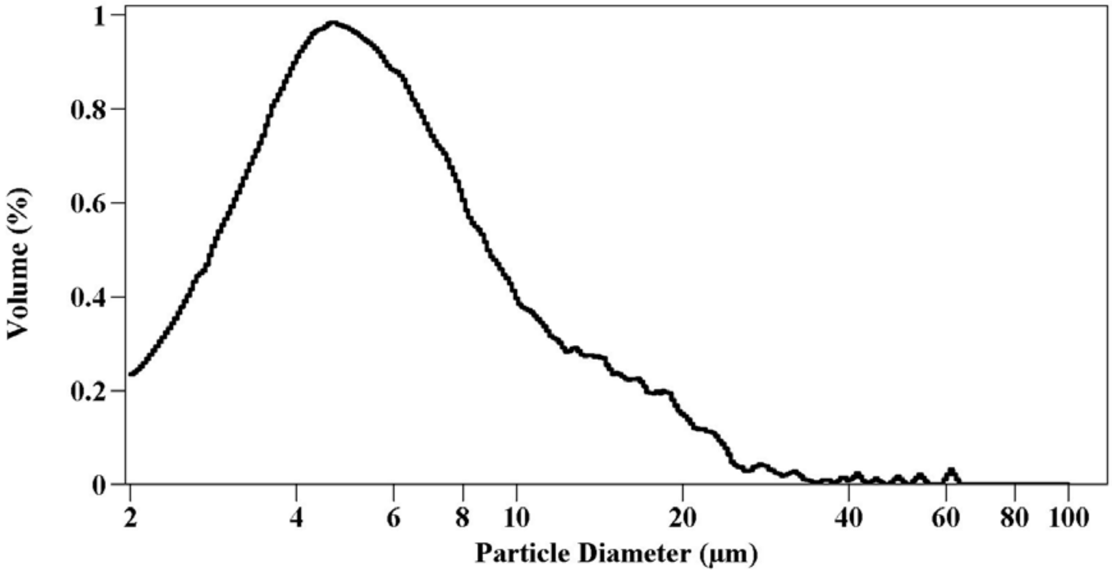

**Figure 177.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from a nylon swab rolled across a Method 201a glass fiber filter obtained from stack sampling conducted on a roller gin lint basket pull system exhaust.

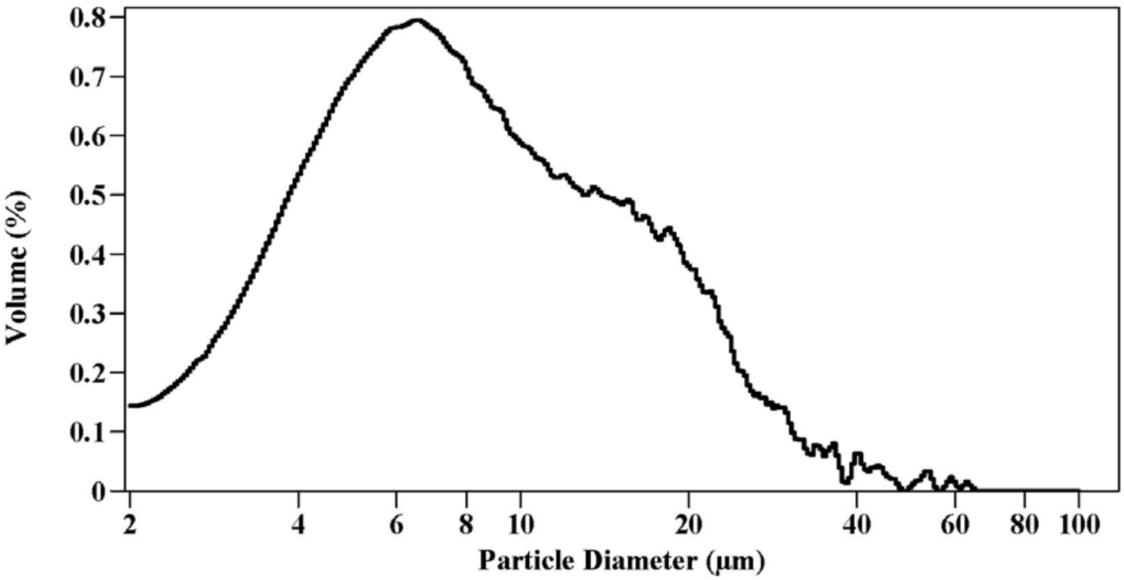

**Figure 178.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the post-cyclone wash obtained from Method 201a stack sampling conducted on a roller gin lint basket pull system exhaust.

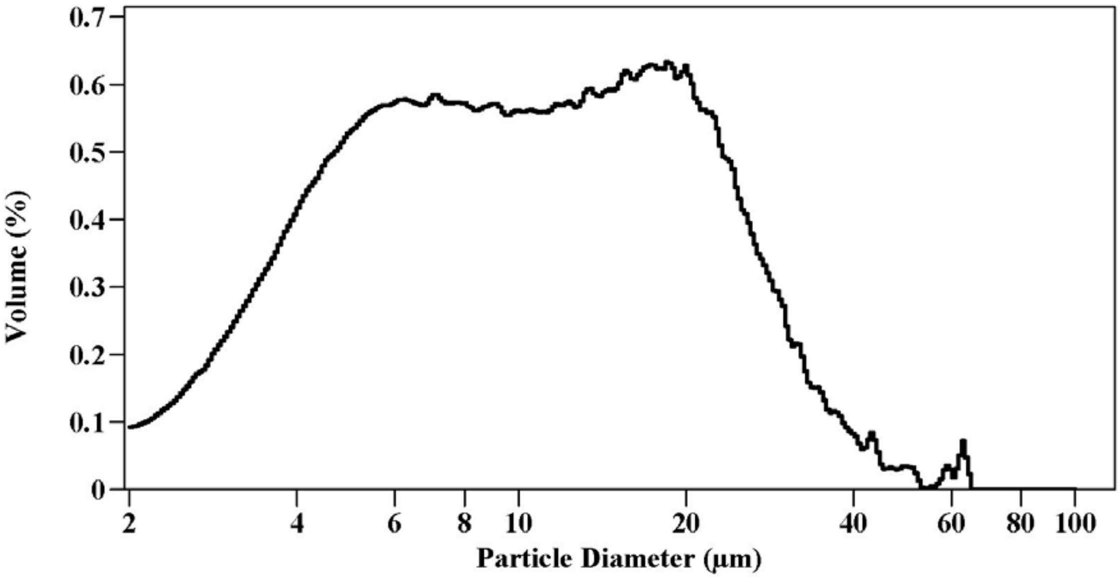

**Figure 179.** Volume based average Coulter PSD (particle diameter in terms of equivalent volume diameter) from the cyclone wash obtained from Method 201a stack sampling conducted on a roller gin lint basket pull system exhaust.

# VITA

Michael Dean Buser
6306 CR 7405
Lubbock, TX 79424

**Education**

Master of Science in Biosystems Engineering
    *Oklahoma State University (December 1997)*
Bachelor of Science in Biosystems Engineering
    *Oklahoma State University (May 1995)*
Associate of Applied Science in Animal Science
    *Rogers State College (May 1990)*

**Professional Experience**

*1998 – Present (Agricultural Engineer)*
United States Department of Agriculture - Agricultural Research Service –
Cotton Production and Processing Research Unit; 1604 E. FM 1294, Lubbock, TX

Responsibilities focused in the area of air pollution engineering, by-product utilization, and gin stand enhancement. Tasks in the area of air pollution have included: developing an ASAE standard for utilizing emission factors to determine emission parameters for cotton gins; abatement device design, development, and implementation; determining the errors associated with EPA $PM_{10}$ and $PM_{2.5}$ samplers; dispersion modeling; evaluation of emitted emissions; and interaction with EPA and state air pollution regulatory agencies on compliance related issues.

Transferring the knowledge base and new technologies to industry is the primary responsibility of research engineers at the USDA. This transfer has been accomplished through 90 plus oral presentations at industry specific conferences and instructing courses for educational workshops and schools. In addition, 35 plus written publications have been published in industry related journals or conference proceedings. Professional activities include: Co-Chairman of the ASAE Cotton Engineering Committee; Secretary of the ASAE Environmental Air Quality Committee; active member of the National Cotton Ginners Association's Air Quality, Gin By-Products, Cottonseed, and Ginning Technology Committees; and Assistant Editor for the Journal of Cotton Science.

**Honors**

Inducted into the Alpha Epsilon, Gamma Sigma Delta, and Phi Kappa Phi honor societies. Received an ASAE Honorable Mention Paper Award in July of 2000. Received Certificate of Recognition as a future leader in Agricultural Engineering from the International Commission of Agricultural Engineering in August of 2002.