Field Evaluation of Portable and Central Site PM Samplers Emphasizing Additive and Differential Mass Concentration Estimates

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Abstract

The US Environmental Protection Agency (EPA) published a National Ambient Air Quality Standard (NAAQS) and the accompanying Federal Reference Method (FRM) for PM_{10} in 1987. The EPA revised the particle standards and FRM in 1997 to include $PM_{2.5}$. In 2005, EPA proposed revisions to this NAAQS to include $PM_{10-2.5}$ but only finalized revisions with a $PM_{2.5}$ FRM and the development of a national monitoring network in 2006. Presently, no EPA designated reference or equivalent method sampler has the ability to directly measure the mass concentrations of PM_{10}, $PM_{10-2.5}$, and $PM_{2.5}$ simultaneously. An additive approach has been used for samplers like the dichotomous monitors to calculate PM_{10} mass concentrations from independent measures of $PM_{10-2.5}$ and $PM_{2.5}$ (i.e. $PM_{10} = PM_{10-2.5} + PM_{2.5}$). A differential approach has been used to calculate $PM_{10-2.5}$ from identical collocated PM_{10} and $PM_{2.5}$ samplers

22 (i.e. $PM_{10-2.5} = PM_{10} - PM_{2.5}$). Since these two approaches have been used widely for PM
23 measurements, it is informative to evaluate their precision and comparability.
24
25 EPA performed collocated tests of five different particle samplers in the Research Triangle Park
26 area of North Carolina to evaluate the comparability and to characterize the additive and
27 differential approaches used to determine particle mass concentrations. The intra-sampler
28 precision of MiniVol, Omni, and dichotomous samplers was less than 8.4%. The precision of
29 PM_{10} measurements using the additive approach with dichotomous samplers was less than 3.5%.
30 The poorest precision of the various $PM_{10-2.5}$ differential approaches was less than 15.1%. No
31 zero or negative $PM_{10-2.5}$ concentrations were calculated using the differential approach. A
32 coefficient of determination of 0.81 or higher was obtained for all paired comparison of $PM_{10-2.5}$.
33 The reported test results show that concentrations calculated from both the additive and
34 differential approaches generally agree among the portable samplers, the more established
35 dichotomous sampler and the Federal Reference Methods.
36
37 *Keywords*: PM_{10}, $PM_{10-2.5}$, $PM_{2.5}$, additive and differential approaches
38
39 **1. Introduction**
40 The Clean Air Act requires EPA to regulate particulate matter (PM) concentrations nationwide.
41 PM is a complex mixture of small solid particles and liquid droplets. The size of particles has
42 been directly linked to their potential for causing health problems (Lippmann et al., 2003). In
43 general, EPA is more focused upon inhalable particles, PM_{10}, that are 10 micrometers (μm) in

diameter or smaller because those particles can affect the heart and lungs and cause serious health effects if inhaled (EPA, 2006a; Zanobetti et al., 2009). In 2006, EPA proposed revisions to the NAAQS for ambient PM (EPA, 2006b). These revisions were made in response to the increasing evidence of adverse health effects associated with human exposure to PM. In recognition that these health effects vary as a function of particle size, the proposed revisions included separate standards for fine particles ($PM_{2.5}$) and coarse particles ($PM_{10-2.5}$), but the latter was not promulgated. $PM_{2.5}$ is the particle size fraction ≤ 2.5 μm in diameter while the $PM_{10-2.5}$ size fraction represents those particles between 2.5 and 10 μm in diameter. A number of studies have identified adverse health effects associated with PM_{10} and $PM_{2.5}$ (Schwartz et al., 1996; Anderson et al., 2005). Recent studies indicate that short-term exposure to ambient $PM_{10-2.5}$ is associated with increased mortality risk and hospital admissions, primarily due to cardiovascular and respiratory effects (Metzger et al., 2004; Houthuijs et al., 2001). Table 1 lists a chronology of EPA's particle pollution standards (http://www.epa.gov/pm/standards.html).

Wilson et al. (1997) reported that the aerodynamic particle diameter for 50 percent penetration into the thoracic airways and lower respiratory tract is 10 μm. PM_{10} is the diameter of particles smaller than 10 μm and divided into $PM_{2.5}$ and $PM_{10-2.5}$ fractions. The two size fractions have different predominant sources and formation mechanisms and consequently different chemical compositions (Brunekeef et al., 2005). $PM_{2.5}$ and $PM_{10-2.5}$ are viewed as separate classes of pollutants and have been measured separately in research and epidemiologic studies. The primary sources of $PM_{2.5}$ are anthropogenic, such as combustion and products of gas to particle conversion. The primary formation mechanisms of $PM_{10-2.5}$ are typically comprised of dust, sand, crushing,

66 grinding and non-exhaust vehicle emissions like tire rubber. $PM_{10-2.5}$ can also consist of biological
67 substances such as pollen and fungi (EPA, 2004; EPA, 2005).
68
69 Three means of determining PM size fractions of interest exist. These include the direct approach
70 where the monitor's design allows for particles of a given size to be captured independent of
71 others. An additive approach is used when one single monitor collect two independent size
72 fractions and then mathematically added together to estimate new fraction. The differential
73 approach involves independent collection of two size fractions which then mathematically
74 subtracted to yield estimates of a third.
75
76 The EPA, state, and local monitoring agencies have maintained a national network to collect and
77 measure PM_{10} since the 1980's and $PM_{2.5}$ since 2000 (EPA, 2000). Dichotomous samplers (dichot)
78 have been used to directly measure the $PM_{2.5}$ and $PM_{10-2.5}$ concentrations since 1980. Using
79 dichotomous data, the additive approach could also be used to estimate the PM_{10} mass
80 concentrations. The differential approach, therefore, could be used to estimate $PM_{10-2.5}$ mass
81 concentrations from collocated PM_{10} and $PM_{2.5}$ data. Several studies have reported on
82 comparisons of $PM_{10-2.5}$ estimates using direct and indirect measurements (Vanderpool et al., 2005;
83 Allen et al., 1999). Findings from these reveal that the additive and differential approaches of
84 mass estimation can produce comparable results to direct measurement for the PM_{10} and $PM_{10-2.5}$
85 size fractions.
86
87 Differential and additive approaches have been used in PM spatial variation studies. In general,

88 the FRM ($PM_{2.5}$ or PM_{10}) and dichot ($PM_{2.5}$ and $PM_{10-2.5}$) are used at central monitoring sites with
89 more portable samplers ($PM_{2.5}$ or PM_{10}) used at satellite sites. Comparing the mass concentrations
90 collected from a central monitor and dispersed portable samplers allows one to assess PM spatial
91 variation across an area. Knowledge of the spatial variation of PM is critical for characterizing
92 potential exposures to particle pollution which can vary greatly from one area to another. $PM_{2.5}$
93 mass is often seen as homogeneous across many urban areas although it varies temporally
94 (Williams et al., 2003). $PM_{10-2.5}$ exhibits more spatial variability and varies substantially across
95 time because it settles by gravity relatively quickly and the sources are not uniformly distributed
96 across an urban area (Ott et al., 2008). Several studies have indicated that $PM_{10-2.5}$ measurements
97 at a single central monitoring site do not provide an adequate measure of the average daily
98 concentration of $PM_{10-2.5}$ across the community compared to that for $PM_{2.5}$ measurement (Moon et
99 al., 2001; Wilson et al., 2005). Chen et al. (2007) and Williams et al. (2000) reported poor spatial
100 agreement for $PM_{10-2.5}$ mass concentrations between a central monitor (dichotomous sampler) and
101 residential outdoor monitoring stations (portable samplers) in the same geographical area.
102 Williams et al. (2008) indicates that the ability of community-based monitors to act as a surrogate
103 for $PM_{10-2.5}$ mass might be problematic. Many factors might have influenced the poor spatial
104 agreement in $PM_{10-2.5}$ measurements reported above. These factors include the disperse nature of
105 $PM_{10-2.5}$ sources, rapid particle settlement, and differences in sampler specifications associated
106 with their deployment (i.e. particle separation technique, flow rate, and sampling height). Any
107 single or combination of these effects could have a major effect on PM mass concentration
108 determinations. There was a lack of information existing as to the comparability of various
109 samplers often used to assess air quality, especially that focusing on the $PM_{10-2.5}$ size fraction.

110 Therefore, we undertook this study to evaluate and compare these two calculation approaches
111 (additive and differential) for both research and FRM type monitors.
112
113 The purpose of this study was to compare two portable samplers (Omni and MiniVol) to the FRM
114 and the dichot and then evaluate the additive and differential approaches for estimating ambient
115 PM. PM_{10} mass concentration calculated from the additive approach using the dichot and $PM_{10-2.5}$
116 mass concentration calculated from the differential approach for the Omni, Minivol, and FRM
117 samplers were compared with direct mass measurements. Precision and comparability of the
118 various instruments to estimate PM_{10}, $PM_{10-2.5}$, and $PM_{2.5}$ were determined and are reported.
119
120 **2. Materials and Methods**
121 *2.1. Study design*
122 To evaluate the performance of the PM samplers, collocated tests were conducted in Research
123 Triangle Park (RTP), North Carolina in 2007. The RTP area was chosen because it served as a
124 convenient (local) method testing site for protocols being developed for future Birmingham
125 (Alabama) studies. The Birmingham studies were designed to determine the spatial and temporal
126 distribution of coarse mode particles within an urban airshed with elevated $PM_{10-2.5}$ levels. The
127 RTP test was designed primarily to ensure functionality of equipment, development of standard
128 operating procedures, and provide operator training for the Birmingham study. The RTP area does
129 not typically experience the levels of $PM_{10-2.5}$ expected for Birmingham and other areas with
130 elevated $PM_{10-2.5}$. Thus, it may not have been an ideal location for the inter-comparison of these
131 samplers and for evaluating the calculation approaches for estimating $PM_{10-2.5}$ levels. Nevertheless,

132 it did provide an opportunity to gather relevant data for comparing these samplers and assessing
133 the implications of the additive and differential approaches on PM estimates.
134
135 Figure 1 shows the sampler placement on the platform with all inlets at the same 5 m elevation
136 above ground level and spaced one meter apart to minimize interference. The 3.3 by 3.3 m
137 auxiliary sampling platform adjacent to the 8 m long motor laboratory has been used for previous
138 EPA studies and provided a safe and secure deployment of the instrumentation (Vanderpool et al.,
139 2005). Three pairs of portable MiniVol (Airmetric, Eugene, Oregon) and Omni (BGI Inc.
140 Waltham, MA) samplers were installed on the monitoring platform. Two Rupprecht and
141 Patashnick dichotomous air samplers (R&P Model 2025 Sampler, Albany, NY), a pair of
142 Graseby-Andersen FRM (Andersen FRM, RAAS-100, Smyrna, GA) and a pair of Rupprecht and
143 Patashnick FRM (R&P FRM, Partisol-FRM 2000, Albany, NY) samplers were also installed on
144 the platform. Each pair of samplers included a PM_{10} and a $PM_{2.5}$ sampler.
145
146 Table 2 lists the specifications of each sampler. Prior to each field test, all samplers were cleaned
147 and passed a leak test. Each sampler was calibrated for volumetric flow rate using a calibrated BGI
148 TriCal. All of the samplers in this study were filter-based employing a 2 μm pore size and 46.2 mm
149 Diameters PTFE filter (Polytetrafluoroethylene, 7592-304, Whatman inc., Florham, NJ). Field
150 blank tests of the samplers were conducted in every sampling event for performance audits. Field
151 blank values were not used to correct the measured PM concentrations but served as means to
152 detect any dramatic filter mass change that would have indicated improper filter handling and
153 transport. No such events occurred. Blank correction is also not permitted per data treatment

154 procedures outlined in the Federal Reference Method for $PM_{2.5}$ (Winter et al., 2004).

155

156 The study consisted of twenty, 45 hour sampling periods. A 45-hour integrated sampling period
157 was used for convenience and to ensure adequate filter mass loading to examine the study
158 hypothesis. Earlier US EPA research (Vanderpool et al., 2006) focusing on determining candidate
159 methods for PM measurement, had identified a range of PM mass concentrations across various
160 U.S. geographical areas. Table 3 now reports summary mass concentration finding from the
161 Vanderpool et al. (2006) effort in comparison to that associated with the current (RTP) data
162 collections. This provides information as to the variability of PM size fraction concentrations
163 observed in North America as it relates to the RTP site. In general, mean $PM_{2.5}$ RTP mass
164 concentrations were very similar those from Gary, Phoenix and Riverside. However, $PM_{10\text{-}2.5}$ and
165 PM_{10} mass concentrations were significantly lower in the RTP area in comparison to those
166 aforementioned cities. The RTP area had a sufficiently low $PM_{10\text{-}2.5}$ and PM_{10} average daily mass
167 concentration that selection of a 24-hour sampling protocol in establishing the current study design
168 might have significantly impacted our ability to fully test the two approaches. Therefore, the
169 selection of an extended 45-hour sampling period was ultimately ensured sufficient data quality to
170 test the study hypothesis. Deviation from the standard 24-hour FRM protocol was not expected to
171 negatively impact sampler operation or the resulting data quality of the size fractionation relative to
172 overall sampler performance. The use of an extended 45-hour sampling period in the current study
173 was viewed as crucial to meet critical data demands (gravimetric analysis) in evaluating the two
174 mass determination approaches, especially those associated with low flow samplers.

175

176 ## *2.2. Sampler Description*

177 ### *2.2.1 Federal Reference Method (FRM)*

178 The FRM is a filter-based, 24 hour sampling period technique with pre- and post- conditioning and
179 weighing of the filter under standardized laboratory conditions. The Andersen FRM and R&P
180 FRM are stand-alone sampling systems meeting national ambient air quality regulations for single
181 day sampling of $PM_{2.5}$ or PM_{10}. The $PM_{2.5}$ FRM samples ambient air through a PM_{10} inlet with a
182 50% cutpoint of 10 μm. The air stream is then directed via the downtubes to the Well Impactor
183 Ninety-Six (WINS) stage with a 50% cutpoint of 2.5 μm (Peters et al., 2001a). For the PM_{10} FRM,
184 the WINS impactor is replaced with an equivalent length of tubing with no size selective
185 characteristics. The nominal flow rate through these FRM systems is 16.67 liter per minute (lpm).
186
187 ### *2.2.2 Dichotomous Sampler*

188 Dichots are designed to provide direct measurements of both $PM_{2.5}$ and $PM_{10-2.5}$ fractions of a
189 PM_{10} aerosol. Ambient aerosol is sampled through a PM_{10} size selective inlet and subsequently
190 divided by a virtual impactor into separate $PM_{10-2.5}$ and $PM_{2.5}$ fractions. These samplers operate at
191 a flow rate of 16.67 lpm with the fine particle flow rate at 15 lpm and coarse particle flow rate at
192 1.67 lpm. Particles in the two resulting flow streams are collected on separate filters for subsequent
193 gravimetric analysis. However, commercially available dichots are inconvenient or impossible to
194 use in remote areas because of their size and power requirements.
195
196 ### *2.2.3 Portable Sampler – Omni and MiniVol Sampler*

197 In 1980, the EPA encouraged state and local air monitoring groups to conduct short-term multi-site

198 pollutant monitoring studies using relatively inexpensive and small portable samplers (Jones et al.,
199 1998). The concept was to "saturate" an area with easily deployable, inexpensive filter-based
200 samplers using battery operation and low flow rates to assess air quality in areas with high
201 concentrations of pollutants or at reclamation sites. Saturation monitoring has also been conducted
202 to characterize the spatial distribution of pollutant concentration or to evaluate source contributions
203 in support of receptor modeling. The Omni and MiniVol samplers are saturation samplers which
204 have been widely used in remote areas lacking AC power. The AC power option was used for all
205 samplers in this study to minimize potential data loss.
206
207 The BGI Omni samplers are lightweight, weatherproof and portable saturation samplers which can
208 operate for up to 48 hours on battery power. A constant volumetric flow rate of 5 lpm was
209 maintained at the system's inlet using calibrated temperature, pressure, and flow sensors. A PM_{10}
210 inlet was used to collect ambient aerosol less than 10 μm on the PTFE filter. A BGI very sharp cut
211 cyclone with a cutpoint of 2.5 μm was installed inline following the PM_{10} inlet for collection of
212 $PM_{2.5}$ mass on the PTFE sampling filter.
213
214 The Airmetric MiniVol sampler is a compact, lightweight, battery-operated monitor, constructed
215 from durable PVC and self-shrouded from weather effects. The MiniVol sampler consists of a
216 size-selective inlet, a flow control unit, a programmable timer, and a rechargeable battery. The
217 PM_{10} inlet was equipped with a PM_{10} impactor. The $PM_{2.5}$ inlet was equipped with a PM_{10} scalping
218 impactor followed by an in-line $PM_{2.5}$ impactor for final size range designation. The standard
219 electrical system for these units was modified to be AC capable for this study. The flow rate of the

220 sampler was 5 lpm. Glisseal high vacuum grease (Lot 130475, borer Chemie, CH-4528 Zuchwil)
221 was used to grease impactor surface on all inlets to eliminate particle bouncing (Baldauf et al.,
222 2001; Tropp et al., 1998; Hill et al., 1999).
223
224 *2.3. Quality Control*
225 MiniVol samplers were examined each sampling day to confirm proper operation. Daily quality
226 control reviews before and after sampling included checks on flow rate, volume of air sampled,
227 elapsed time, air leaks, and unusual filter conditions. The Omni, FRM, and dichot samplers
228 required little maintenance. They performed automated calibrations for volumetric flow rate
229 involving ambient temperature, filter temperature, and ambient pressure measurements. All
230 samplers were audited for nominal flow rates prior to and immediately after sampling using a BGI
231 TriCal Volumetric Air Flow Calibrator. Samplers were expected to maintain nominal flow rates ±
232 5% of initial settings to be considered valid. All filter-based sampling was performed using a single
233 lot of filters respective to size and media type. The MiniVol sampler's impactor was cleaned and
234 recoated after every two sampling periods (90 hours). The impaction well of the FRM WINS was
235 replaced after every two sampling periods. Although the impactors was recoated and the WINS
236 replaced after every 90 hours of sampling, they had minimal observable build-up upon inspection
237 which would have potentially increased particle rebound effects and resulting decreases in sampler
238 precision. Consequently, the selection of the extended 45-hour sampling period had no obvious
239 negative impact on sampler performance.
240
241 Gravimetric mass analysis of filters prior to and following exposure was conducted in an

242 environmentally controlled balance room. This balance room was an enclosed chamber where
243 temperature and relative humidity (RH) were made constant. The normal set-points were 21.5 ±
244 0.5^0C and 35 ±5% RH, in conformity with current Federal requirements for PM filter mass
245 determination. The filters were placed in the balance chamber for a minimum of 24 hour
246 undisturbed to allow for filter equilibration. Gravimetric analysis was then performed at least twice
247 for each filter. In order to verify the gravimetric analysis, 100% replicate weighing (with a ±5 μg
248 reweigh threshold) was used for each filter during all pre- and post- weighing operations. The
249 average mass was then used to calculate concentrations.
250
251 **3. Results and Discussion**
252 *3.1. Precision*
253 The precision of each sampler was based upon comparisons of collocated duplicate samplers.
254 Precision of dichot sampler was defined as the (A-B)*100/(A+B) when N=2. The "A" and "B"
255 represent the observed mass concentration of the sampler replicates. Precision of the MiniVol and
256 Omni were defined as the average of abs((A-m)/m) and abs((B-m)/m) when N=3 where m is the
257 median value. Table 4 shows the average precision and general statistics calculated from the
258 equations above associated with 20 sampling events for the dichot, MiniVol, and Omni samplers.
259 The results indicate that the precision error of direct measurements of PM mass in three size
260 fractions was less than 8.4% from all samplers. The precision of dichot PM_{10} measurements using
261 the additive approach was less than 3.5%. The precision of $PM_{10-2.5}$ measurements determined by
262 the differential approach was less than 15.1%.
263

264 The precision of the FRMs cannot be reported because only one $PM_{2.5}$ and one PM_{10} FRM were
265 operated in this study. But in previous studies, Peters et al. (2001B) reported precision error of
266 $PM_{2.5}$ FRMs to be 2 to 6%. Wilson et al. (2002) indicated that $PM_{2.5}$ and PM_{10} FRMs provide
267 relatively precise (±10%) methods for determining the mass of material remaining on a Teflon
268 filter after equilibration. Chow et al. (2008) indicated that well prepared collocated samplers could
269 have precision of approximately 10% for a variety of $PM_{2.5}$ FRM samplers.
270
271 Wilson et al. (2002) indicated that the $PM_{10-2.5}$ precision error ranged from a few percent to as
272 much as 40%. Allen et al. (1999) measured $PM_{10-2.5}$ precision of 15% for the Harvard Impactor
273 using the difference between independent PM_{10} and $PM_{2.5}$ monitors. They also indicated that the
274 precision for $PM_{10-2.5}$ measurements may be three or four times worse than the associated $PM_{2.5}$ or
275 PM_{10} data, depending on the measurement approach, $PM_{2.5}$ to PM_{10} ratio, and $PM_{10-2.5}$
276 concentrations.
277
278 *3.2. Mass Concentration Comparison in Three Size Fractions*
279 Table 5 shows the average mass concentrations (AMC) measured from 20 sampling measurements.
280 Data reporting mass concentrations associated with each of the various monitors are reported. The
281 use of the additive or differential approach is indicated as appropriate. Discussion about unique
282 findings for the various combinations are reported below.
283
284 *3.2.1 PM_{10} (all direct except dichot being additive)*
285 The comparison of AMC determined by indirect measurement to direct measurement was very

286 similar. The AMC of two portables sampler was the same as the Andersen FRM but about 4% less
287 than the R&P FRM. The AMC of the dichot calculated using the additive approach was 1% higher
288 than the AMC of the two FRMs. The PM_{10} dichot concentration calculated from the additive
289 approach was 3% higher than the concentrations measured directly from the MiniVol and Omni
290 samplers.
291
292 *3.2.2 $PM_{2.5}$ (all direct measurement)*
293 The AMC of the MiniVol and Omni portable samplers were 9% higher than the AMC of the dichot
294 and FRMs samplers. The AMC of the dichot was the same as the Andersen FRM but 2% higher
295 than the R&P FRM. The AMC of the MiniVol was 4% higher than the Omni samplers.
296
297 *3.2.3 $PM_{10-2.5}$ (all but dichot being differential)*
298 The results show significantly more variability in the resulting $PM_{10-2.5}$ mass concentrations as
299 compared to the PM_{10} or $PM_{2.5}$ estimates. The poorest $PM_{10-2.5}$ AMC agreement observed was that
300 from the two portable samplers which represented only 72% of that provided by the dichot. The
301 AMC collected from the two FRMs deviated by approximately 10% in comparison to the dichot.
302 Lower flow rate samplers (5 lpm) had an AMC for this size fraction that was less than the higher
303 flow rate samplers (16.67 lpm) regardless of the approach involved. Even so, samplers operating at
304 the same flow rate were sometimes observed to yield AMC differences greater than 10%. In
305 general, the direct measurement approach routinely provided higher AMCs as compared to those
306 produced by the differential approach.
307

308 Even for the direct measurement of $PM_{10-2.5}$ from the dichot sampler, there still exists some
309 uncertainty because of its complicated analysis of the $PM_{10-2.5}$ concentration. Allen et al. (1999)
310 reported that the dichots have substantial $PM_{10-2.5}$ error due to a decrease in $PM_{10-2.5}$ measurement
311 precision when $PM_{2.5}$ to $PM_{10-2.5}$ ratios are high, changes in sampler characteristics as particles
312 deposit on inside surfaces, or the loss of coarse particles from filters during transport. Some
313 particle losses within the dichot have been reported (Marple et al., 1990).
314
315 The Coefficient of Determination (R^2) of three PM size fractions had been reported in Tables 6 – 8.
316 R^2 is used to indicate the strength of a relationship between two inter-samplers. R^2 values of at
317 least 0.95 for all the PM_{10} and $PM_{2.5}$ comparisons were obtained. R^2 of $PM_{10-2.5}$ between and
318 among samplers having the same flow rate were also investigated. Between the two portable (i.e.
319 MiniVol and Omni) samplers, the R^2 value was 0.87. For the more stationary samplers, the R^2
320 value of $PM_{10-2.5}$ between dichot vs. Andersen FRM, dichot vs. R&P FRM, and Andersen FRM vs.
321 R&P FRM were 0.82, 0.87 and 0.86, respectively. The R^2 values between portable and stationary
322 samplers ranged from 0.81 to 0.92. An R^2 of 0.92 was obtained for the best paired comparison of
323 $PM_{10-2.5}$ in this study (R&P FRM vs MiniVol or Omni).
324
325 **4. Conclusion**
326 This study provided the means to directly test the comparability of vastly different approaches
327 (direct, additive, differential) in estimating various PM size fractions mass concentrations. Such
328 testing was needed to determine the comparability of data being produced in studies carried out
329 throughout the world involving the different approaches and often involving highly different

sampling instrumentation. All of the various approaches or instrumentation available to researchers have their own specific advantages and disadvantages that might influence the scientific community-at-large in decisions to deploy them in future research studies. The findings presented here represent a first attempt in defining some of the issues that need to be considered when researchers select any combination of these various mass estimation instrumentation approaches.

Reproducible (< 15.1% precision) mass concentrations for the $PM_{10-2.5}$ size fraction would appear to be capable of being generated in environmental monitoring studies regardless of the selection of any approach or instrumentation type evaluated as part of this study. Even so, precision error of this particular size fraction was often two to five times poorer than that established for the other two size fractions even when sampling periods double that of the normal daily monitoring periods were evaluated. Shorter time periods (e.g., 24-hour) would be suspected of having even higher precision errors due to lower mass loadings negatively impacting the resulting gravimetric analyses. It is suspected that numerous factors associated with the individual mass estimation and instrumentation type selections evaluated here resulted in the higher cumulative $PM_{10-2.5}$ precision errors.

Comparability of mass concentrations within the $PM_{2.5}$ or PM_{10} size fractions were extremely well correlated ($R^2 > 0.95$) regardless of the approach or instrumentation selected. Conversely, the $PM_{10-2.5}$ size fraction exhibited slightly poorer mass concentration agreement ($R^2 > 0.81$) under various combinations of test conditions. Both the additive and differential approaches would appear to yield highly similar mass estimates for the $PM_{10-2.5}$ size fraction regardless of

instrumentation type. Even so, low flow rate monitors (MiniVol and Omni) would however, appear to yield mass concentration estimates slightly lower for this size fraction in comparison to those like the higher flow rate FRMs. Therefore, the state-of-the science at this point would indicate researchers can be fairly confident that mass estimates they obtain regardless of technique selection are generally representative of the environment being monitored. However, enough variability was exhibited between the various test combinations that would indicate common instrument deployment (same monitor type and selection of mass estimation procedures) across all spatial settings would be advantageous in obtaining $PM_{10-2.5}$ mass estimates of the highest degree of uniformity.

Acknowledgements

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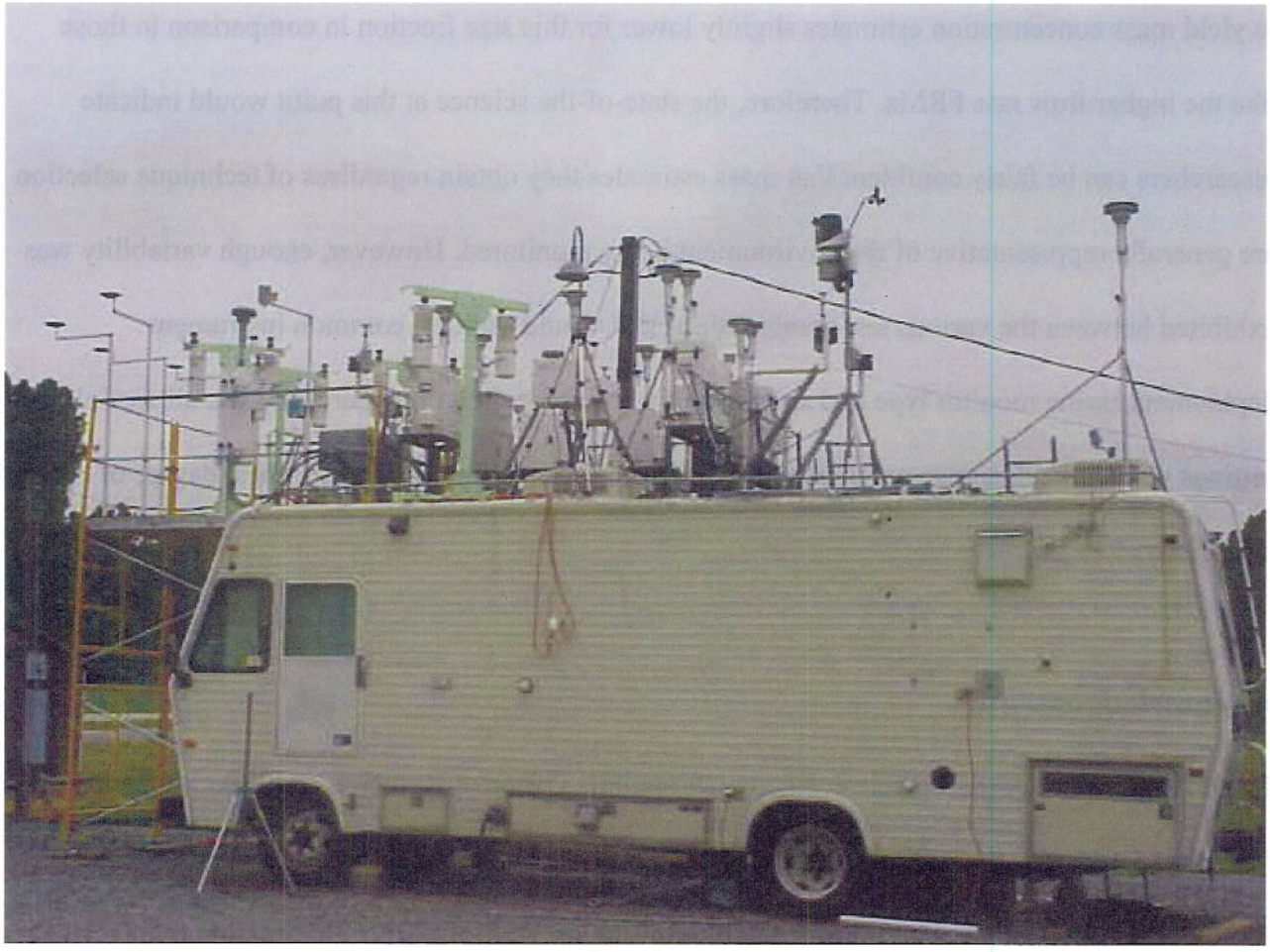
374 Figure 1. Samplers set up on the platform at field site (RTP, NC, 2007)



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384 Table 1. Particle pollution standards, (http://www.epa.gov/pm/standards.html)

Date	National Ambient Air Quality Standards	24 h average ($\mu g/m^3$)	Annual ($\mu g/m^3$)
1971	Total Suspended Particles (TSP)	260	75
1987	PM_{10}, particles size smaller than 10 micrometers (replaced TSP)	150	50
1997	Added $PM_{2.5}$, particles size smaller than 2.5 micrometers, standards	35	15
2006	Proposed $PM_{10-2.5}$ standards	70	N/A

Table 2. Specifications of tested Samplers

Sampler	Fraction Collected	Additive or Differential approach	Flowrate, lpm
R&P Dichot	$PM_{10-2.5}$, $PM_{2.5}$	$PM_{10}=PM_{10-2.5} + PM_{2.5}$	16.67
Anderson FRM	PM_{10}, $PM_{2.5}$	$PM_{10-2.5} = PM_{10}-PM_{2.5}$	16.67
R&P FRM	PM_{10}, $PM_{2.5}$	$PM_{10-2.5} = PM_{10}-PM_{2.5}$	16.67
Airmetric MiniVol	PM_{10}, $PM_{2.5}$	$PM_{10-2.5} = PM_{10}-PM_{2.5}$	5.0
BGI Omni	PM_{10}, $PM_{2.5}$	$PM_{10-2.5} = PM_{10}-PM_{2.5}$	5.0

Table 3. Mass concentration range and mean ($\mu g/m^3$) from the Vanderpool et al. (2006) locations and this study.

	$PM_{2.5}$	$PM_{10-2.5}$	PM_{10}
Gary, IN. 2003 March - April	10.3 – 46.9 Mean = 22.8	4.5 – 58.1 Mean = 19.9	22.6 – 85.0 Mean = 42.6
Phoenix, AZ. 2003 May - June	6.4 – 22.0 Mean = 11.0	26.5 – 209.0 Mean = 55.7	37.1 – 230.9 Mean = 66.7
Riverside, CA. 2003 July - August	9.9 – 32.7 Mean = 17.7	16.2 – 46.1 Mean = 30.4	27 – 69.3 Mean = 48.0
Phoenix, AZ. 2004 January - February	6 – 22.4 Mean = 13.2	7.7 – 95.1 Mean = 39.5	14.8 – 177.5 Mean = 52.8
Phoenix, AZ. 2005 April - May	4.9 – 16.6 Mean = 12.8	23.4 – 122.8 Mean = 46.2	30.1 – 134.6 Mean = 56.0
RTP, NC. 2007 April - May	5.2 – 22.4 Mean = 12.8	3.1 – 14.6 Mean = 8.0	10.7 – 30.8 Mean = 20.8

Table 4. General statistics of the Intra-Sampler Precision (%) of MiniVol, Omni and Dichot from 20 sampling events.

Size Fraction		MiniVol, N=3	Omni, N=3	Dichot, N=2
PM_{10}	Mean	2.3	6.2	3.5*
PM_{10}	Max.	8.6	18.1	12.6
PM_{10}	Min.	0.6	1.1	0.3
$PM_{2.5}$	Mean	4.6	5.7	3.6
$PM_{2.5}$	Max.	11.5	12.9	20.0
$PM_{2.5}$	Min.	0.5	1.7	0.0
$PM_{10\text{-}2.5}$	Mean	13.2**	15.1**	8.4
$PM_{10\text{-}2.5}$	Max.	42.1	28.0	21.0
$PM_{10\text{-}2.5}$	Min.	1.3	5.3	1.2

* additive approach ** differential approach

Table 5. General mass concentration ($\mu g/m^3$) statistics (N=20) of the collocated samplers.

	Dichot	Andersen FRM	R&P FRM	MiniVol	Omni
PM_{10}, Mean	21.2 ± 5.5*	20.5 ± 5.8	21.4 ± 5.9	20.5 ± 6.1	20.6 ± 5.5
PM_{10}, Max.	30.8*	29.3	30.8	30.7	32.2
PM_{10}, Min.	12.4*	10.7	11.7	12.2	11.9
$PM_{2.5}$, Mean	12.5 ± 4.2	12.5 ± 4.5	12.2 ± 4.4	13.8 ± 4.7	13.2 ± 4.2
$PM_{2.5}$, Max.	20.2	20.5	19.9	22.4	21.6
$PM_{2.5}$, Min.	5.5	5.2	5.2	7.1	6.3
$PM_{10\text{-}2.5}$, Mean	8.7 ± 3.2	7.9 ± 3.1**	9.2 ± 3.4**	6.6 ± 2.7**	7.4 ± 2.7**
$PM_{10\text{-}2.5}$, Max.	14.0**	14.3**	14.6**	11.6**	14.3**
$PM_{10\text{-}2.5}$, Min.	4.9**	3.3**	5.4**	3.1**	4.2**

* additive approach ** differential approach

Table 6. Coefficient of Determination, R^2, for PM_{10} Fraction Comparison

All P-values < 0.0001

	Omni	Dichot (Additive)	Andersen FRM	R&P FRM
MiniVol	0.97	0.97	0.98	0.97
Omni		0.95	0.98	0.98
Dichot (Additive)			0.96	0.95
Andersen FRM				0.96

Table 7. Coefficient of Determinations, R^2, for $PM_{2.5}$ Fraction Comparisons

All P-values < 0.0001

	Omni	Dichot	Andersen FRM	R&P FRM
MiniVol	0.97	0.98	0.98	0.99
Omni		0.96	0.96	0.98
Dichot			0.97	0.98
Andersen FRM				0.98

Table 8. Coefficient of Determinations, R^2, for $PM_{10-2.5}$ Fraction Comparisons

All P-values < 0.0001

	Omni	Dichot	Andersen FRM	R&P FRM
MiniVol	0.87	0.87	0.81	0.92
Omni		0.83	0.86	0.92
Dichot			0.82	0.87
Andersen FRM				0.86

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