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Supplement of

Successes and challenges of measuring and modeling atmospheric mercury at the part per quadrillion level: a critical review

M. Sexauer Gustin et al.

Correspondence to: M. Sexauer Gustin (mgustin@cabnr.unr.edu)

13 ***Lumex***

14

15 After startup, the Lumex instrument baseline will drift downward asymptotically for 1 to 3
16 hours before leveling off at a stable value, after which a meaningful zero baseline value can be
17 set. Laboratory tests show after stabilization, a mean blank concentration of $-0.13 \pm 0.34 \text{ ng m}^{-3}$
18 ³ can be maintained for at least 3 hours while operating in the internal zero reference mode,
19 with no indication of baseline drift (linear regression coefficient of 0.00004). External source
20 injections of GEM in zero air at calculated concentrations of 240, 120, and 60 ng m^{-3} resulted in
21 mean recoveries of 99.7, 97.9, and 95.9 percent, respectively. If the recovery efficiency trend is
22 extrapolated to 1 ng m^{-3} the Lumex would under report by about 15%.

23

24 ***Sheu and Mason (2001) methods and detailed results***

25

26 The filter pack consisted of two Teflon filters in front of three cation exchange membranes in-
27 series. They assumed the first Teflon filter collected PBM and the second was a blank, the first
28 cation exchange membrane collected GOM, the second was a blank, and the third kept GOM
29 from diffusing from the pump. They slightly altered the mist chamber from the design of
30 Lindberg and Stratton (1998).

31

32 The mist chamber collected ~34% more GOM than the cation exchange membrane. Sheu and
33 Mason suggested this may be an artifact of the mist chamber in marine air. The denuders in
34 general indicated higher concentrations than the filter pack measurements. They suggested this
35 was due to an inherent blank problem or due to GEM being oxidized at the KCl surface. They
36 tested whether there were reactions in the mist chamber when exposed to light and found this
37 did not occur. PBM was also measured on Teflon filters and concentrations ranged from <9 to
38 147 pg m^{-3} .

39

40 ***Cation exchange membrane***

41

42 The original material used for the cation exchange membrane “ICE 450” has been discontinued
43 and replaced with a polyethersulfone membrane (“Mustang S”). In laboratory and field tests
44 collection of GOM in ambient air were not significantly different; however, in the field the ICE
45 450 in general had higher collection efficiency, although they were not significantly different,
46 suggesting it may collect some forms of GOM that the Mustang S does not (Huang and Gustin,
47 submitted). Recent field tests indicated there was no significant loss when surrogate surfaces
48 and UNR passive sampler for GOM concentrations (box sampler) were deployed for 2 to 4
49 weeks (Huang and Gustin, submitted). In addition, use of an automated analytical system
50 (Tekran 2600) for membrane analyses significantly improved the method detection limit (Huang
51 and Gustin, submitted).

52

53 ***TGM vs GEM comparison***

54

55 Limited field work has indicated that the Tekran® 2537 alone does collect all TGM with a soda
56 lime and particulate filter in-line. Temme et al. (2002) compared data from two Tekran
57 systems in Antarctica, one measuring TGM using Tekran® 2537 only and the other using the
58 Tekran® 1130/1135/2537 to measure GEM, GOM and PBM. They suggested that GOM and GEM
59 were collected as TGM by the 2537. Based on 7 months of data from Mercury Deposition
60 Network site MDN 98 in Nevada, Gustin (2011) found the sum of the GEM and GOM was more
61 similar to TGM than GEM alone. This comparison was in an environment with relatively dry air
62 and similar comparisons in humid air may yield more comparable concentrations since humidity
63 can reduce GOM collected on surfaces to GEM.

64

65 ***Detailed discussion on the potential for different forms in the air in Florida***

66

67 There are several important points not presented in these papers. First in Peterson
68 et al. (2012), the discrepancy between modeled dry deposition using a multi-resistance
69 model, and measured deposition varied by location and across different seasons. This
70 supports the contention that there are different forms in air. At these sites modeled
71 deposition was weakly correlated with measured deposition ($r^2 = 0.24$ and 0.28 $p < 0.5$, and
72 $r^2 = 0.16$ $p < 0.10$). This indicates at times there were GOM compounds in the air that the
73 Tekran® system was more efficiently capturing, and at other times this measurement was
74 less efficient, or the model was not accurately capturing deposition.

75 The constant model parameterization and similar environmental conditions across
76 all sites (cf. Peterson et al., 2012) provides a basis for comparison of measured deposition
77 across all sites. In all cases measured deposition was significantly higher (2 to 18 times)
78 than modeled values (Figure SI 1). At all sites there was a pattern in the ratio of
79 measured:modeled deposition with a decline in the summer and increase in fall.

80 Differences between the measured and modeled values were similar for all sites in the
81 winter. The Davie Site near Fort Lauderdale (DAV) site had the lowest ratio of measured:
82 modeled deposition during the other seasons, suggesting the denuder was more efficient at
83 collecting GOM compounds at this location, or that the model was better at capturing
84 deposition at this site. Ratios were highest at Tampa (TPA) in the spring and fall relative to
85 the other sites. At Outlying Landing Field (OLF), the most background site, measured:
86 modeled values were higher in the spring (time periods when long range transport occurs)

87 and in the fall, a time of year when mobile source pollution would be higher. An alternate
88 explanation is that the model bias varies seasonally because of errors in meteorology,
89 surface/land type, PBL height, etc. and this could explain some of the variability in the
90 measured:modeled ratio. For a detailed discussion of how these data demonstrate there
91 are different forms of GOM in the air see the SI.

92 Data of Peterson et al (2012), presented in Figure SI 2, shows that the surrogate
93 surface-derived deposition velocities are highest for the TPA site that is primarily impacted
94 by mobile sources, and overlap for DVE (local point source impact and mobile source
95 impacted), and were intermediate for OLF (most background site with a long range
96 transport impact). Despite the fact these were calculated using the Tekran® GOM data,
97 since there were similar conditions across the sites, this also indicates there are different
98 forms in the air at these sites. Dry deposition measured by the surrogate surface will be
99 influenced by the deposition velocity or the chemical compound. Lin et al. (2006), using the
100 model of Wesely (1989), reported the deposition velocity of HgO was 2-fold higher than
101 that for HgCl₂. Thus, if there are different forms in the air, data collected by the surrogate
102 surface will vary across space and time.

103
104 If we assume:

105 1- the cation exchange collects GOM quantitatively then the collection efficiency of KCl-
106 coated denuder (in a Tekran 1130) for different GOM compounds is HgBr₂ (1.6)>HgSO₄
107 (2.3)=HgCl₂ (2.4)>HgO (3.7) >Hg(NO₃)₂ (12.6) (Values in parenthesis represent concentrations
108 measured by the cation exchange membrane: concentrations reported by the Tekran® denuder
109 using the laboratory manifold system in zero air); and

110 2- based on the work presented in Gustin et al. (2012) that deposition associated with
111 natural background is 0.23 ng m⁻² h⁻¹ (corrected using the 0.2 ng m⁻² h⁻¹ factor); deposition
112 associated with oxidation of GEM due to mobile source emissions at TPA, DAV, and OLF was
113 0.3, 0.3, and 2.3 ng m⁻² h⁻¹, respectively; deposition from long range transport was 2.8 ng m⁻² h⁻¹
114 at all sites in the spring; and deposition at DVE associated with direct emission or formation
115 due to associated oxidants was 0.3 ng m⁻² h⁻¹, there could be different sources associated with
116 each specific source. For example, natural background could be due to reactions with Br and
117 perhaps S compounds; deposition associated with mobile sources could be Hg-O or Hg-N
118 associated compounds; that due to long range transport in the spring could be HgO related or
119 HgBr₂ compounds; and that associated with local point sources could be, Hg-S, Hg-O or Hg-N
120 related compounds (cf. Timonen et al., 2013). Since collection efficiency for HgSO₄ is relatively
121 good, this might explain the better agreement between the measured and modeled deposition
122 at DVE and this site is impacted by oil electrical generation facilities. These ratios could also be
123 influenced by ozone and humidity (see Lyman et al., 2010; McClure et al. 2014; Huang and
124 Gustin, this issue, and discussion below). The atmospheric constituents at all sites are more
125 complicated than that in the laboratory manifold (discussed below). Recent work by Huang et
126 al. (this issue) based on thermodesorption of the nylon membranes, at OLF there are different
127 compounds depending on the time of the year.

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129
130

131 **Detailed discussion of RAMIX data**

132

133 Ambient air RM concentrations measured during RAMIX by the DOHGS were
134 typically higher than the Tekran® derived RM. Although data are noisy, RM was typically
135 higher as measured by the DOHGS at midnight to late morning (up to 350 pg m⁻³), relative
136 to the Tekran® system. At 12:00, values measured by the DOHGS compared with the
137 Tekran® systems were higher by ~75 pg m⁻³. The variation in the discrepancy between
138 DOHGS and Tekran® RM measured over 24 h points to different GOM compounds in the air
139 and production of GOM in the manifold.

140 Another piece of evidence for different GOM compounds in the air and points to fast
141 reactions that were hypothesized to occur in the manifold, is presented in Figure SI 3 from
142 Gustin et al. (2013). The timing of the hourly peaks in GOM as measured are similar to
143 those observed for ozone (one at 1200 h and another later in the afternoon). The first is
144 attributed to mixing of air pollution to the surface from the free troposphere. The second is
145 due to a reaction with local pollution (see Gustin et al., 2013). This latter could be due to
146 the reaction suggested by Calvert and Lindberg (2005) for Hg and O₃:

147 $GEM + O_3 \rightarrow HgO_3 \text{ (unstable)} \rightarrow OHgOO \rightarrow O_2 \text{ and } HgO$. Equation 2.

148 They also indicated that HgO₃ would deposit on aerosols and other surfaces where it may
149 form HOHgOH, and this state Hg could be transformed into a variety of Hg (II) compounds
150 with other ions. For additional discussion of atmospheric chemistry see the discussion
151 below.

152 Thus, the RAMIX experiment demonstrated that the Tekran® KCl-coated denuder
153 measurement for GOM was biased low, there were different chemical forms in the air, and
154 reactions were occurring in the manifold (Gustin et al., 2013). The latter has implications
155 for reactions that could occur in ambient air. Although there were some limitations with
156 each method applied during the experiment, the project also demonstrated: 1) using GEM
157 spikes, measurements made by two carefully calibrated Tekran® 2537 units could
158 significantly differ and the only way to explain this was a discrepancy between the
159 instruments; 2) GOM was significantly underestimated by the denuder (17 to 24%
160 recovery of HgBr₂ spikes in ambient air in Reno); and 3) the DOHGS measured different RM
161 forms than the Tekran® KCl-coated denuder.

162

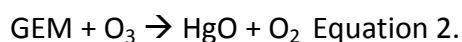
163

164 **Detailed discussion of atmospheric chemistry**

165

166 Hynes et al (2009) summarized the laboratory work that has been done on the Hg + O₃
167 reaction by Hall (1995), Pal and Aryia (2004a and b), Summer et al.(2005), Rutter et al. (2012).
168 All of these studies found O₃ to participate in a reaction with GEM. Based on an analyses of the
169 electronic configuration, Tossell (2003), and Shepler and Peterson (2003) indicated that HgO
170 is a weakly bound molecule. Hynes et al (2009) calculated a rate coefficient for the following
171 equation:

172



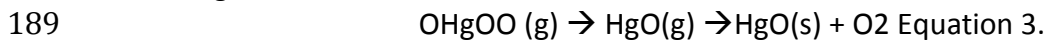
173

174 They indicated the rate coefficient was unreasonably high and suggested this reaction
175 was unlikely to occur by homogenous process. Hynes et al (2009) noted that if the Hg + O₃
reaction does exist it must occur as a weakly bound complex; and "In laboratory experiments

176 such a complex could diffuse to the reactor surface to form solid mercuric monoxide, possibly
177 by way of oligomer formation. Such a process would be energetically favorable because of the
178 exothermicity of oligomer formation and the lattice energy associated with formation of the
179 solid.”

180 Peterson et al (2009) indicated that the rate coefficient for O₃ production developed by
181 Pal and Ayria (2004a) was sufficient to produce 5 to 10 pg of HgO per hour; and using the rate
182 coefficient developed by Pal and Ayria (2004b) and a concentration of OH of 1 x 10⁶ molecules
183 per cm³ (Seinfeld and Pandis, 1996) approximately 1 to 10 pg of HgO could be produced per
184 hour.

185 Ariya et al. (2009) suggested that most atmospheric oxidation reactions occur by
186 multiple steps complicating our ability to experimentally study and calculate reaction rates, and
187 understand reactions. They proposed after the initial reaction between GEM and O₃ that the
188 following occurred;



190 The last two steps are irreversible. Additional work by the Ariya group showed that the O₃
191 oxidation rate increases in the presence of CO and relative humidity (Snider et al., 2008).

192

193 ***PBM detailed information***

194

195 Keeler et al (1995) used Teflon and quartz fiber filters to measure PBM in the Great
196 Lakes area and showed PBM had a bimodal distribution. A paper by Kim et al. (2012) showed a
197 bimodal size distribution with the strength of this distribution varying by season and location,
198 and PBM was measured for 0.18 to 18 μm using an eight-stage MOUDI (Micro Orifice Uniform
199 Deposit Impactors, Midel M 100R, MSP Corp) with Zeflour filters (Pall corp.) and a cascade
200 impactor with glass fiber filters. Particles across all size ranges were measured.

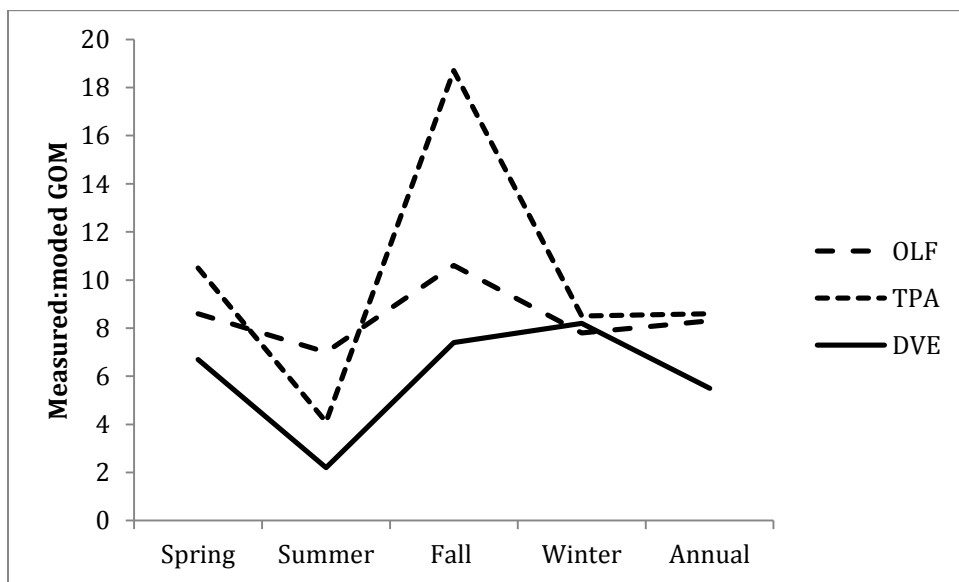
201 Keeler et al. (1995) reported 60-100% of PBM was found in fine particle fraction in
202 Detroit. A significant fraction of PBM can be found in coarse particle at coastal sites (Engle et al.,
203 2008; Malcolm et al., 2007). Kim et al. (2012) found PBM in the winter in Chuncheon and Seoul
204 Korea were in the < 2.5 μm fraction, however, in the summer, in Seoul, there was a constant
205 distribution across all particulate sizes, and in Chuncheon, concentrations in the summer were
206 higher in the 2 to 10 μm fraction. This indicates the grain size fraction measured by the
207 Tekran® is not adequate due to the need for an elutricator to keep particles of > 2.5 μm from
208 entering the denuder.

209 Rutter et al. (2008) used quartz fiber filters and KCl-coated quartz fiber filters in-series
210 to collect PBM and GOM and found that concentrations on the first filter were higher, and the
211 latter comparable to that measured with a Tekran® system. It is possible some GOM was
212 collected on the quartz fiber filter. PBM measured by the quartz filter was 2.7 times higher
213 than that measured by the Tekran® 1135. Since RM is semi-volatile (Rutter and Schauer, 2007),
214 Rutter et al. (2008) suggested that since the particulate module is maintained at 50oC, RM
215 could be lost. Rutter et al. (2008) suggested that evaporative loss could be augmented by the
216 presence of other semivolatile compounds such as NH₃NO₃ or other semivolatile organics.
217 Malcolm and Keeler (2007) observed less PBM collected on quartz filters for 12 hrs versus 4 hrs
218 and suggested a negative sampling artifact that could be due to reduction due to reaction with
219 gases in the air (such as O₃) or relative humidity.

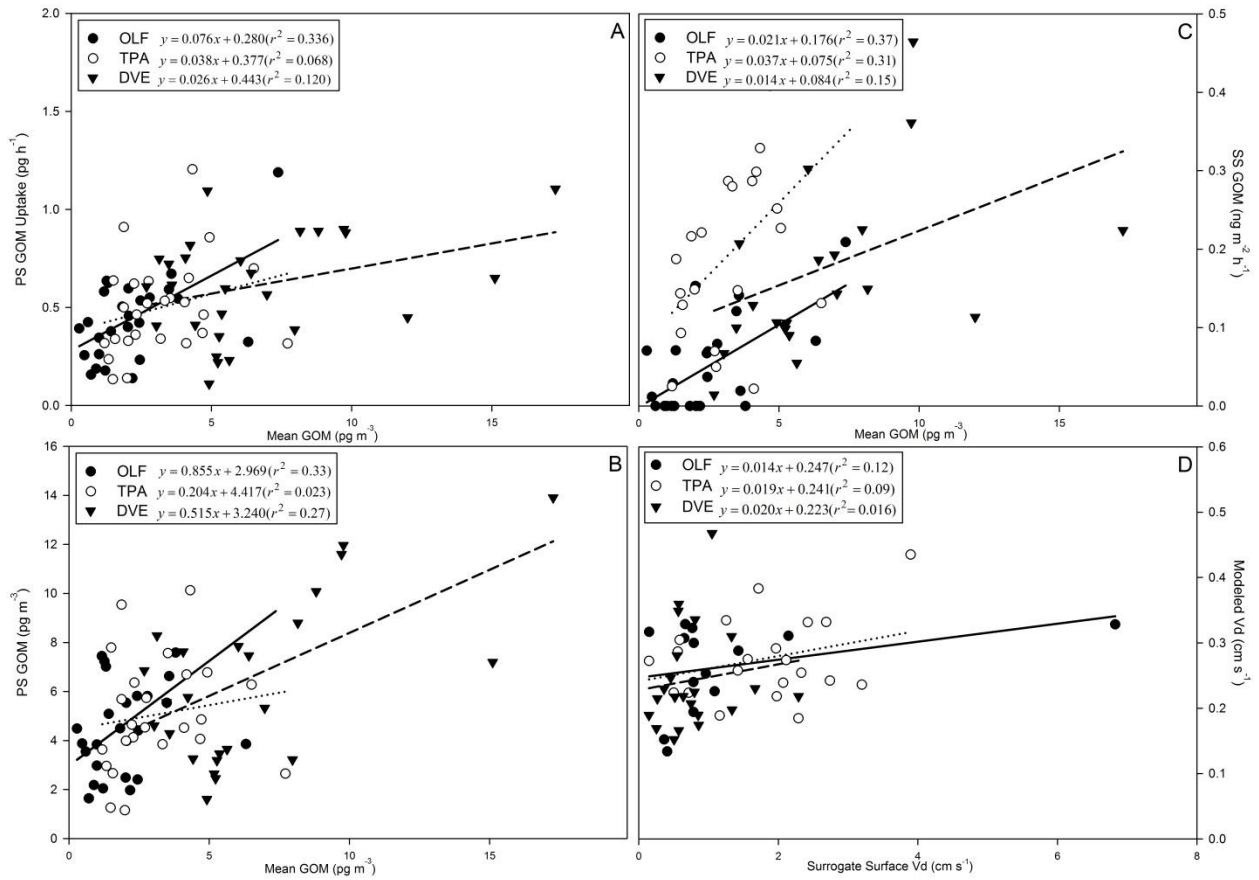
220 Talbot et al. (2011) measured PBM with a Millipore® 1 µm fluoropore membrane to
221 compare with data collected using the Tekran® 1135 unit and found the filter method collected
222 21% higher PBM and 85% of the data disagreed by 25% or more. In addition, the Tekran®
223 measurement had a high percentage of data below the detection limit while the particulate
224 filter (average blank concentration 25 pg and blank for the Tekran® measurement was 0 pg m⁻³)
225 always had detectable Hg. The filter method was more responsive to pollution episodes
226 associated with a combustion source. It is also possible their filters were collecting some GOM.
227

228 ***Details on the sampling configuration in Weiss-Penzias et al. 2003***

229
230 For this study they describe GEM only concentrations being measured from May 2001
231 to May 2002 by a 2537; GOM measurements began in September 2001 by the 1130; and PBM
232 was only measured at selected times using filters. For the GEM measurement the instrument
233 was housed in a trailer with a black plastic covered line heated to 40°C provided by Tekran and
234 a quartz filter to remove PBM and GOM. The quartz filter likely worked for PBM and possibly
235 some forms of GOM. Once installed the 1130 was configured to collect a sample over 4 hours.
236 Quartz filters were used to collect PBM over 27 hours on 47 mm quartz fiber filters. The
237 samplers collected particulate matter < 2.5 µm, and 2.5 to 10 µm.
238

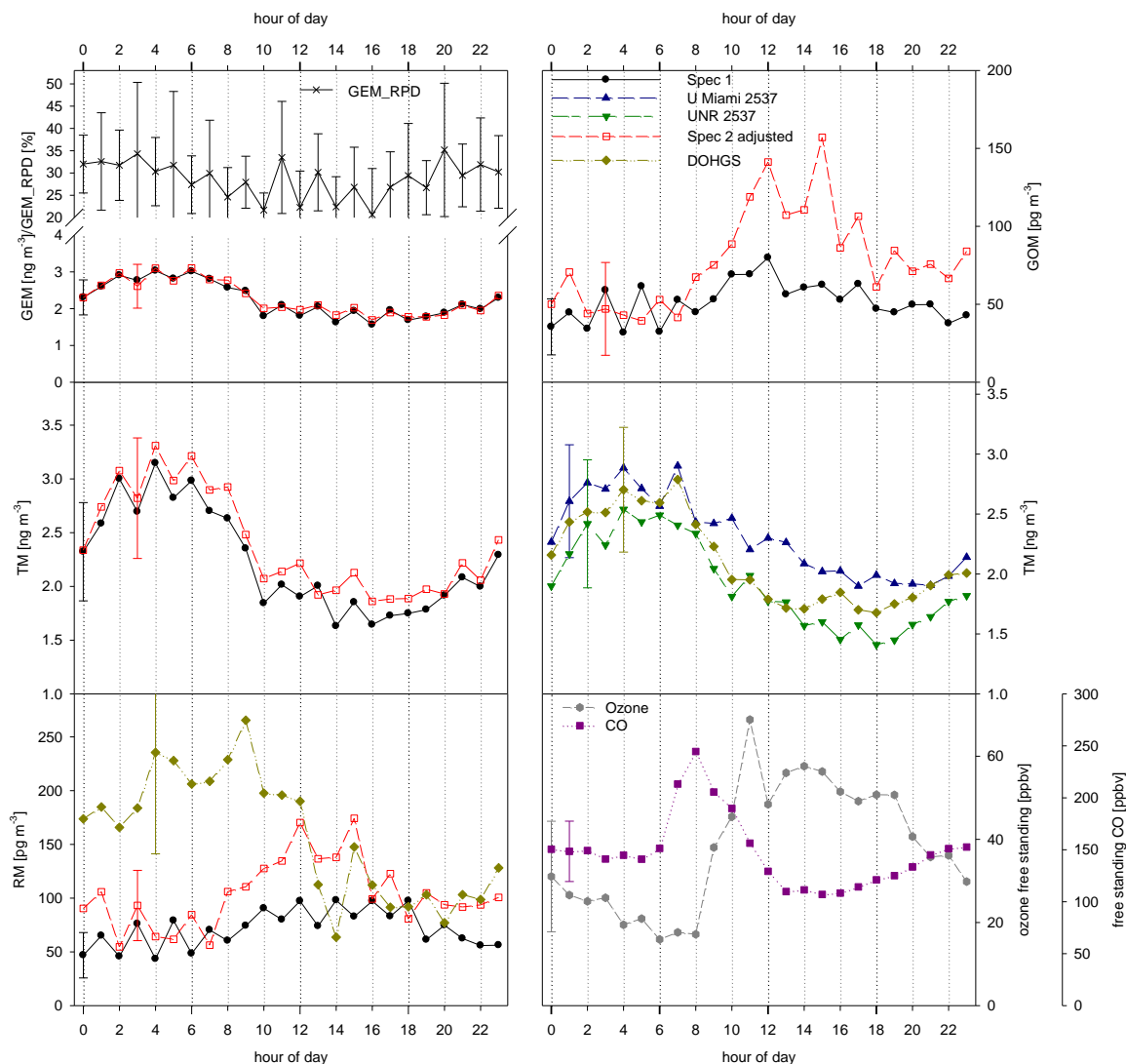


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240
241 Figure SI 1. Ratio of surrogate surface measured (uncorrected dry deposition value-see section
242 3.2.2) to modeled values of GOM dry deposition for Spring (March-May), Summer (June-August),
243 Fall(September-November), winter (December-January), and for the year. Data were collected
244 from July 2009 to July 2010 at Outlying Landing Field (OLF) near Pensacola, a site just east of
245 Tampa (TPA), and one near Fort Lauderdale (DVE) and the town of Davie (Data from Peterson et al.,
246 2012).
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Figure SI 2. Figure 4 from Peterson et al (2012). Reprinted with permission from Peterson et al., 2012, Copyright 1 May 2012, Elsevier Limited.



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254 Figure SI 3. Figure 2 from Gustin et al (2013). Reprinted with permission from Gustin et al., 2013,
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256

257

258

259 **References cited**

260 Ariya, P. A., Peterson, K., Snider, G., Amyot, M.: Mercury chemical transformation in the gas,
261 aqueous heterogenous phases: State-of – the art science and uncertainties, in: Mercury
262 Fate and Transport in the Global Atmosphere: Emissions, Measurements and Models,
263 Pirrone, N., Mason, R., (Eds.), Springer, Dordrecht, NL, 459-502, 2009.

264

265 Calvert, J.G., Lindberg, S.E.: Mechanisms of mercury removal by O₃ and OH in the atmosphere,
266 Atmos. Environ., 39, 3355-3367, 2005.

267

268 Engle, M.A., Tate, M.T., Krabbenhoft, D.P., Kolker, A., Olson, M.L., Edgerton, E.S., et al.:
269 Characterization and cycling of atmospheric mercury along the central US Gulf Coast,
270 *App. Geochem.*, 23, 419-437, 2008.
271

272 Gustin, M. S.: Exchange of Mercury between the Atmosphere and Terrestrial Ecosystems, in:
273 *Environmental Chemistry and Toxicology of Mercury*, Liu, G., Cai, Y., O'driscoll, N., (Eds.),
274 John Wiley and Sons, Hoboken, New Jersey, 423-452, 2011.
275

276 Gustin, M.S., Weiss-Penzias, P.S., and Peterson, C.: Investigating sources of gaseous oxidized
277 mercury in dry deposition at three sites across Florida, U.S.A., *Atmos. Chem. Phys.*, 12,
278 9201-9219, 2012.
279

280 Gustin, M.S., Huang, J., Miller, M.B., Peterson, C., Jaffe, D.A., Ambrose, J., et al.: Do We
281 Understand What the Mercury Speciation Instruments Are Actually Measuring? Results
282 of RAMIX, *Environ. Sci. Technol.*, 47(13), 7295-7306, 2013.
283

284 Hall, B.: The gas-phase oxidation of elemental mercury by ozone, *Water Air Soil Poll.*, 80, 301-
285 315, 1995.
286

287 Huang, J., Gustin, M.S. Impacts of relative humidity on GOM measurements, submitted to
288 *EST*
289

290 Huang, J., Gustin, M.S.: Use of passive sampling methods and models to understand sources of
291 mercury deposition to high elevation sites in the Western United States, 49 (432-441)DOI
292 10.1021/es502836w,2015.
293
294

295 Hynes, A.J., Donohoue, D.L., Goodsite, M.E., Hedgecock, I.M.: Our current understanding of
296 major chemical and physical processes affecting mercury dynamics in the atmosphere
297 and at the air-water/terrestrial interfaces, in: *Mercury Fate and Transport in the Global*
298 *Atmosphere: Emissions, Measurements and Models*, Pirrone, N. and Mason, R. (Eds.),
299 Springer, Dordrecht, NL, 427-458, 2009.
300

301 Keeler, G., Glinsorn, G., and Pirrone, N.: Particulate mercury in the atmosphere: Its significance,
302 transport, transformation and sources, *Water Air Soil Poll.*, 80, 159-168, 1995.
303

304 Kim, P.R., Han, Y.J., Holsen, T.M., and Yi, S.M.: Atmospheric particulate mercury: Concentrations
305 and size distributions, *Atmos. Environ.*, 61, 94-102, 2012.
306

307 Lin, C-J., Pongprueksa, P., Lindberg, S.E., Pehkonen, S.O., Byun, D., Jang, C.: Scientific
308 uncertainties in atmospheric mercury models I: Model science evaluation, *Atmos.*
309 *Environ.*, 40, 2911-2928, 2006
310

311 Lindberg, S.E., and Stratton, W.J.: Atmospheric mercury speciation: Concentrations and
312 behavior of reactive gaseous mercury in ambient air, *Environ. Sci. Technol.*, 32, 49-57,
313 1998.
314

315 Lyman, S.N., Gustin, M.S., and Prestbo, E.M.: A passive sampler for ambient gaseous oxidized
316 mercury concentrations, *Atmos. Environ.*, 44, 246-252, 2010.
317

318 Malcolm, E.G. and Keeler, G.J.: Evidence for a sampling artifact for particulate-phase mercury in
319 the marine atmosphere, *Atmos. Environ.*, 41, 3352-3359, 2007.
320

321 McClure, C.D., Jaffe, D.A., and Edgerton, E.S.: Evaluation of the KCl Denuder Method for
322 Gaseous Oxidized Mercury using HgBr₂ at an In-Service AMNet Site, *Environ. Sci.*
323 *Technol.*, 48 (19), 11437-11444, 2014.
324

325 Pal, B. and Ariya, P.A.: Studies of ozone initiated reactions of gaseous mercury: kinetics, product
326 studies, and atmospheric implications, *Phys. Chem. Chem. Phys.*, 6, 572-579, 2004a.
327

328 Pal, B. and Ariya, P.A.: Gas-phase HO[•]-Initiated reactions of elemental mercury: Kinetics,
329 product studies, and atmospheric implications, *Environ. Sci. Technol.*, 38, 5555-5566,
330 2004b.
331

332 Peterson, C., Alishahi, M., Gustin, M.S.: Testing the use of passive sampling systems for
333 understanding air mercury concentrations and dry deposition across Florida, USA, *Sci.*
334 *Total Environ.*, 424, 297-307, 2012.
335

336 Peterson, C., Gustin, M., and Lyman, S.: Atmospheric mercury concentrations and speciation
337 measured from 2004 to 2007 in Reno, Nevada, U.S.A., *Atmos. Environ.*, 43, 4646-4654,
338 2009.
339

340 Rutter, A.P., Hanford, K.L., Zwiers, J.T., Perillo-Nicholas, A.L., Schauer, J.J., Olson, M.L.:
341 Evaluation of an offline method for the analysis of atmospheric reactive gaseous
342 mercury and particulate mercury, *J. Air Waste Manag. Assoc.*, 58, 377-383, 2008.
343

344 Rutter, A. P., and Schauer, J. J.: The impact of aerosol composition on the particle to gas
345 partitioning of reactive mercury, *Environ. Sci. Technol.*, 41, 3934-3939, doi:
346 10.1021/es062439i, 2007.
347

348 Rutter, A.P., Shakya, K.M., Lehr, R., Schauer, J.J., Griffin, R.J.: Oxidation of gaseous elemental
349 mercury in the presence of secondary organic aerosols, *Atmos. Environ.*, 59, 86-92,
350 2012.
351

352 Seinfeld, J.H., and Pandis, S.N.: *Atmospheric Chemistry and Physics*, John Wiley and Sons, New
353 York, New York, 1998, 1203 pp.
354

355 Shepler, B.C. and Peterson, K.A.: Mercury monoxide: A systematic investigation of its ground
356 electronic state, *J. Phys. Chem. A.*, 107: 1783-1787, 2003.
357

358 Sheu, G.R. and Mason, R.P.: An examination of methods for the measurements of reactive
359 gaseous mercury in the atmosphere, *Environ. Sci. Technol.*, 35, 1209-1216, 2001.
360

361 Snider, G., Raofie, F., and Ariya, P.A.: Effects of relative humidity and CO(g) on the O(3)-initiated
362 oxidation reaction of Hg(0)(g): kinetic & product studies, *Phys. Chem. Chem. Phys.*, 10,
363 5616-5623, 2008.
364

365 Summer, A. L., Spicer, C. WW., Satola, J., Mangaraj, R., Cowen, K. A., Landis, M. S., Stevens, R.
366 K., Atkeson, T. D.: Environmental chamber studies of mercury reactions in the
367 atmosphere, in: *Dynamics of Mercury Pollutin on Regional and Global Scales*, N. Pirrone
368 and K. R. Mahaffey (Eds.), 193-212, 2005.
369

370 Talbot, R., Mao, H.T., Feddersen, D., Smith, M., Kim, S.Y., Sive, B., et al.: Comparison of
371 Particulate Mercury Measured with Manual and Automated Methods, *Atmosphere*, 2,
372 1-20, 2011.
373

374 Temme, C., Einax Jr., W., Ebinghaus, R., and Schroeder, W.H.: Measurements of Atmospheric
375 Mercury Species at a Coastal Site in the Antarctic and over the South Atlantic Ocean
376 during Polar Summer, *Environ. Sci. Technol.*, 37, 22-31, 2002.
377

378 Timonen, H., Ambrose, J. L., and Jaffe, D. A.: Oxidation of elemental Hg in anthropogenic and
379 marine airmasses, *Atmos. Chem. Phys.*, 13, 2827-2836, 2013.

380 Tossell, J.A.: Calculation of the energetics for oxidation of gas-phase elemental Hg by Br and
381 BrO, *J. Phys. Chem. A*, 107, 7804-7808, 2003.

382 Wesely M.L.: Parameterization of surface resistances to gaseous dry deposition in regional-scale
383 numerical models. *Atmos. Environ.* 23, 1293-1304, 1989.

384
385